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Modelling of Hydrological and Non-Point Source Pollution Regimes in Big Creek Watershed

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MODELLING OF HYDROLOGICAL AND NON-POINT SOURCE POLLUTION REGIMES IN BIG CREEK WATERSHED

By
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ABSTRACT

The hydrological and non-point source loading processes of the Big Creek Marsh and Big Creek Watershed were investigated in this study. The Big Creek Watershed in south-western Ontario was modelled with AnnAGNPS (Annualized AGricultural Non-point Source). The AnnAGNPS model was first calibrated and validated with observed streamflow data in the neighbouring Canard River Watershed. Nash-Sutcliffe model efficiencies for monthly streamflow predictions were 0.75 and 0.72, for the calibration and validation periods. In the Big Creek Watershed the north-eastern and south-eastern regions were found to produce the highest sediment and nutrient loads. A water budget model for the Big Creek Marsh was developed to investigate hydrologic historic processes in the wetland. In the model assessment three potential wetland operating plans were reviewed and compared to the observed pumping data. A sensitivity analysis of the water budget model was performed. An investigation of Lake Erie's influence on the Marsh was also included.

DEDICATION

This thesis is dedicated to my family and friends,
that supported me during my research.

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LIST OF ABBREVIATIONS

Abbreviation	Long Name
AET	Actual Evapotranspiration
AMSL	Above Mean Sea Level
AnnAGNPS	Annualized Agricultural Non-Point Source
ANSI	Area of Natural and Scientific Interest
ANSWERS-2000	Areal Nonpoint Source Watershed Environment Response Simulation - 2000
ARC	Antecedent Runoff Condition
CN	Curve Number
CREAMS	Chemicals, Runoff, and Erosion from Agricultural Management Systems
DEM	Digital Elevation Map
ERCA	Essex Region Conservation Authority
ESA	Environmentally Significant Area
ET	Evapotranspiration
GIS	Geographic Information System
GLEAMS	Groundwater Loading Effects on Agricultural Management Systems
HRU	Hydrologic Response Unit
HSPF	Hydrological Simulation Program - Fortran
HUSLE	Hydro-geomorphic Universal Soil Loss Equation
IBA	Important Bird Area
NPS	Non-point Source
NRCS	Natural Resources Conservation Services

OMAFRA	Ontario Ministry of Agriculture, Food and Rural Affairs
PET	Potential Evapotranspiration
PTTW	Permit to Take Water
RUSLE	Revised Universal Soil Loss Equation
SCS	Soil Conservation Service
SWAT	Soil and Water Assessment Tool
USDA	United States Department of Agriculture
USGS	United States Geological Survey
USLE	Universal Soil Loss Equation
WB	Water Budget
WRP	Wetland Reserve Program

CHAPTER 1:INTRODUCTION

1.1 Statement of the Problem

The management of water quality and quantity is a vital concern for human health and the health of natural systems. Water quality fundamentally influences both aquatic communities and aquatic predictors (ERCA, 2008). Characterizing, monitoring, and understanding surface water quality is essential to management of natural systems connected to rural, urban or industrial land. Developing an understanding of were, how, and when water moves is also essential to the management of natural systems. In 2008 a study was initiated by the Township of Amherstburg, Essex Region Conservation Authority (ERCA), and other local stakeholders to generate a better understanding of the Big Creek Watershed's water quality and water quantity characteristics. The results of the study will be utilized to influence decisions and policies within the watershed.

The Big Creek Watershed is located in Essex County in south-western Ontario, has a drainage area of approximately 70 km^2 and outlets to Lake Erie. The drainage area of the watershed includes parts of the urban core of the Township of Amherstburg, but the landuse cover is primarily agricultural. The main tributary of the Big Creek Watershed drains directly to the Big Creek Marsh. The Marsh is a riparian wetland where the water levels have been artificially managed since 1909 by the landowners using a system of pumps and a control dam (Ducks Unlimited Canada, 2007).

The Big Creek Marsh is the largest natural heritage feature within the watershed consisting of nearly 9 km^2 of Provincially Significant Wetland: in addition, the wetland is a provincially identified significant life science Area of Natural and Scientific Interest (ANSI), an Environmentally Significant Area (ESA), and a globally Important Bird Area (IBA) (ERCA, 2008). Since European settlement the urbanization and clearing of natural features in the Big Creek Watershed has substantially altered the drainage. This in turn has increased flow rates, decreased water quality and diminished the ecological function of the watershed (Waldron, 1998). Several rare plant species reside in Big Creek Marsh including the American Lotus, Prairie White Fringed-orchid and Swamp Rose Mallow, in addition to, numerous rare animal species found in the Marsh including the Fox Snake, the Prothonotary Warbler and the Bald Eagle (Wilson and Cheskey, 2000).

To protect and manage the natural features within the watershed and the water bodies to which the watershed drains the creation of a comprehensive Watershed Plan was initiated. Several local stakeholders are involved with the development of the Watershed Plan as a proactive tool in planning and developmental activities (ERCA, 2011a). The stakeholders involved with the Watershed Plan include land owners of the marsh area, citizens living in the watershed, the Town of Amherstburg, Ministry of Natural Resources, Ministry of Environment, and the Essex Region Conservation Authority (ERCA).

The watershed planning process includes the assessment of the natural resources in the watershed and the establishment of appropriate strategies for the management of these features and processes under present and future conditions. The Watershed Plan focuses on three major areas: water quantity, water quality and natural heritage. The Big Creek Watershed Plan must recognize the importance of creating an inclusive vision for the watershed which supports a vibrant agricultural industry, a wide variety of recreational opportunities, with plentiful opportunities for community growth (ERCA, 2011a).

This research work is undertaken to assist and in conjunction with the Big Creek Watershed Plan. It will incorporate two major components. The first component is the modelling of the hydrological functions of the watershed's most significant natural feature, the Big Creek Marsh. The second component is the modelling of the NPS (non-point source) pollution generated over the watershed drainage area. The results of the modelling will aid in the development of the Big Creek Watershed Plan.

Understanding the hydrological function of a wetland is fundamental to its planning process, rehabilitation, and management (Erwin, 2009). Understanding the historic baseline water levels and flows is fundamental to make appropriate decisions for any planning process involving a wetland. The ecological health of a wetland system is highly dependent on the hydrological functions (WRP, 1997). The water budget (WB) development for Big Creek Marsh will inform planners of the historic hydrological functions of the wetland.

Both sediments and nutrients transported via agricultural runoff have been identified as primary sources of NPS pollution. NPS pollution can be defined as pollution that is generated over an area that cannot be tracked to a single point. NPS pollutants that accumulate in water bodies can create water quality problems (NOAA, 2007). The benefits that can result from the tracking, monitoring, simulating, and controlling NPS pollutants include improvement in water quality, ecological rehabilitation, and recreational benefits. When developing a watershed plan considering remedial measures, it is crucial for planners to evaluate the consequences of these actions (Oogathoo, 2006). The AnnAGNPS model has been successfully utilized as a NPS modelling tool several times in the southern Region Ontario (Jayasuriya, 2007; Das et al., 2007; Das et al., 2008).

1.2 Objectives of the Study

There are two primary objectives of this study. The first objective is to model the hydrological and NPS loading in the Big Creek Watershed with the AnnAGNPS model. The second major objective of this study is to investigate and quantify the hydrological processes within Big Creek Marsh. To achieve the two major objectives the following sub-objectives will be completed:

- Examining the applicability of AnnAGNPS model for the watersheds in Essex region.

- Calibrating and validating the AnnAGNPS model in the adjacent Canard River Watershed.
- Verifying the AnnAGNPS model in the Big Creek Watershed with the limited observed streamflow data in the watershed.
- Identifying the relative areas that are generally more susceptible to soil erosion within the Big Creek Watershed.
- Assessing the agricultural nutrient loadings within the Big Creek Watershed.
- Developing a WB model for estimating both the natural and the anthropogenic hydrological processes within Big Creek Marsh.
- Testing the applicability of the Big Creek Marsh WB model with the available observed data.
- Reviewing the three potential operating scenarios of the wetland with the WB model.
- Conducting a sensitivity analysis of the Marsh WB model and investigating the Marsh's relationship with Lake Erie.

1.3 Structure of the Thesis

This thesis research study is composed of seven chapters to meet the objectives outlined in the previous section. A brief description of the seven chapters is outlined below.

Chapter 1 provides an introduction to the existing problem in the Big Creek Watershed and Big Creek Marsh. This chapter also describes the specific objectives of the research study.

Chapter 2 outlines an extensive literature review for the thesis study. This chapter focuses on four major areas of research. The first part of this chapter provides a general review of WB studies and WB modelling. The second section of this chapter provides an in-depth review of wetland WBs and WB modelling. The third and fourth sections review NPS loading and NPS loading models with an emphasis on studies pertaining to AnnAGNPS.

Chapter 3 provides a summary of the major concepts and processes simulated in AnnAGNPS. This chapter provides a brief review of the model's development, input methodology, and the equations used to simulate the watershed's NPS loadings.

Chapter 4 describes the development of the Big Creek Marsh WB model. The model was designed to approximate the hydrological processes within the wetland. The chapter discusses the major logic assumptions both in the model structure. The data utilized in the development of the WB's model construction is also outlined.

Chapter 5 briefly describes the development of the AnnAGNPS input database. A description of the AnnAGNPS input database's calibration and validation in a watershed neighboring Big Creek, the Canard River Watershed, is outlined. Limited observed streamflow data from the Big Creek Watershed is reviewed for model verification. The results of the Big Creek Watershed AnnAGNPS simulation are summarized and discussed.

Chapter 6 presents the results of the Big Creek Marsh WB model investigation over the forty year study period. In the chapter a review of the three potential operating wetland phases are considered and compared to the available observed pumping data. A model sensitivity analysis is undertaken and an investigation of Lake Erie's effect on the Marsh is outlined.

Chapter 7 presents the conclusions derived from the results of the wetland and watershed simulation studies. Recommendations from each study are also outlined in this chapter.

CHAPTER 2:LITERATURE REVIEW

2.1 Introduction

The following chapter provides literature review for the research work. This chapter will focus on four major sections: a general review of water budget (WBs), a review of wetland WBs, a review of non-point source (NPS) pollution models, and a review of AnnAGNPS studies.

2.2 Water Budgets

A WB is a term used to quantify the various components of the hydrologic cycle (Rahman, 2007). The hydrologic cycle (or the water cycle) describes how water moves through the atmosphere, on and under the earth's surface, and though vegetation (USGS, 2011). A WB is a systematic measure of the individual hydrologic inflows, hydrologic outflows, and storage within a selected control boundary. In general WB analysis reviews the hydrologic processes on a variety of temporal and spatial scales. WBs follow the principal of the conservation of mass and in their simplest form can be expressed as outlined in Equation 2-1.

$$Change\ in\ Storage = Water\ Inflows - Water\ Outflows \qquad \text{Equation 2-1}$$

When performing a WB several components must be selected including a control volume, inflows and outflows which are deemed significant, and appropriate physical and empirical models that emulate their respective components. As each WB is unique to temporal and spatial scales, individualized components of inflows and outflows may need to be considered; however, in general a standard set of flows are normally considered. The common components of a WB include precipitation, evapotranspiration (ET), surface water movement, and groundwater movement (Healy et al., 2007). ET is a term representing both the effects of evaporation and transpiration. Evaporation is the phase change from a liquid to a gas releasing water from a wet surface into the air and transpiration is the phase change when water is released into the atmosphere by vegetation (Ritter, 2006). The ET component of a WB generally represents a large portion of the annual WB and is one of the more difficult components to calculate directly (Healy et al., 2007 and Ritter, 2006).

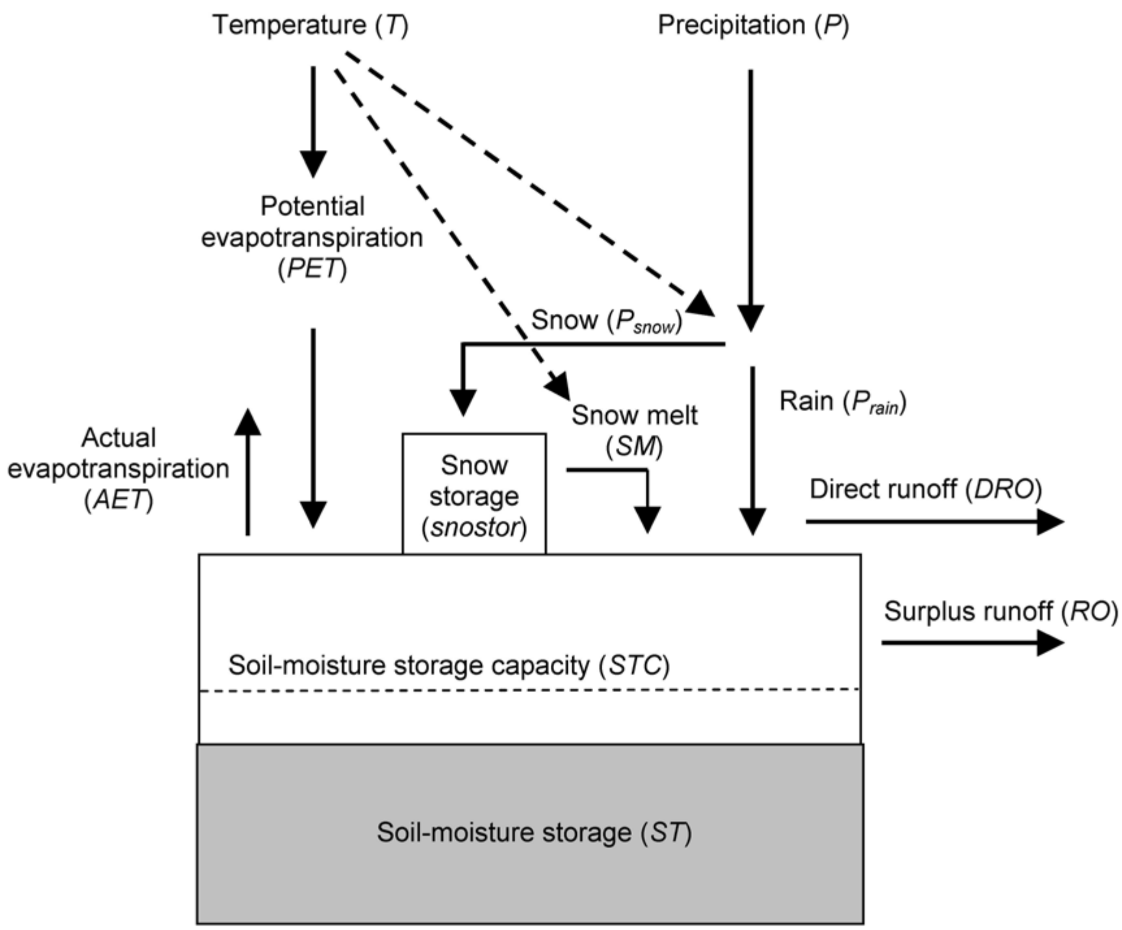

Figure 2-1: Water Budget Diagram

(Source: McCabe and Markstrom, 2007)

WBs provide a foundation for evaluating water's relationship and influence on ecological, social and economic systems (Conservation Ontario, 2010). A WB analysis of a system or region can provide insight into how human activities affect water supply and the environment. WBs are closely linked to other fundamental cycles of energy and chemical transfer on, under, and above the earth’s surface (Healy et al., 2007). Therefore analyzing WBs can provide significant insights into water quality issues.

The benefits and advantages in undertaking and analyzing a WB are closely linked to addressing major decisions, including (Conservation Ontario, 2010; MNR, 2008; and CCL, 2001):

- providing general insight to hydrological processes
- estimating the amount of water flowing through a watershed
- understanding the processes and pathways of the water
- determining the reliability of the water supply to plan sustainably
- highlighting key factors that may limit the reliability of these water supplies
- identifying significant groundwater recharge areas
- planning for landuse and watershed resources, including documentation
- ensuring sustainable development
- determining the receiving stream capacity for waste discharge
- assessing risk exposure
- evaluating economic benefits to the community
- reporting environmental conditions and status

WBs are simple in concept, as illustrated in Figure 2.1, but because of the complexity of natural systems are difficult to quantify (NOAA, 2010). WBs quantify the continuous and intricate movement of water under, on and above the earth’s surface.

2.3 Wetland Water Budgets

The following section will review WBs in wetlands. Wetlands are often difficult to define and lack an internationally and universally accepted definition. The lack of a perfect fit definition is due to the great variability between geographical regions, including meteorological patterns, land characteristics, soil properties, local vegetation, and streamflow regimes (Environment Canada, 1997). Definitions created by interested political and scholastic organizations require that all or some of the following be found in a wetland: periodic or seasonal flooding, the presence of a surface or near surface ground water table, hydric undrained soil, and the growth of hydrophytes (wetland vegetation) are promoted (Tiner, 1999). With the term wetland having such a broad and often elusive definition, a generic methodology appropriate for WBs in these lands are consequently are also difficult to outline because of the same variability.

The hydrologic processes that occur in wetlands are essentially the same that occur outside of wetlands. The major components of the hydrologic cycle that should be considered are the same as other WBs. The components include surface water flow, precipitation, groundwater flow, and ET (USGS, 1999): however, unlike other WBs, for a wetland to exist both a favourable geological setting and an adequate supply of water are necessary. The Wetland Reserve Program (WRP, 1993a, b) supported by the United States Department of Agriculture (USDA) Natural Resource Conservation Service has produced a small series of literature that agree with the USGS report (1999) about the components of a WB that should be considered in a wetland study. In addition to the listed WB flows, coastal wetlands could also be affected by wave action (USGS, 1999). Wind, wave or tidal action can cause movement of sand barriers along shorelines and drastically alter the storage in a wetland.

2.3.1 Wetland Evapotranspiration

Evapotranspiration (ET) is one of the major losses in wetlands. The ET in a wetland can be greater than precipitation as was found by Mitsch's and Reeder's (1992) WB in a Great Lake coastal wetland. In prairie wetlands summer ET is greater than precipitation (Hayashi and Van Der Kamp, 2009). The complexity of ET makes the estimate of this WB component difficult to quantify.

The following section contains a review of wetland ET. The ET that actually occurs in wetlands is a function of several variables, including the available water and atmospheric conditions. The maximum potential rate of ET, potential evapotranspiration (PET), is the amount of water that would be evaporated under an optimal set of conditions, including an unlimited supply of water (Ritter, 2006). However, the actual evapotranspiration (AET) rate is the amount of water that is transpired and evaporated.

Factors that influence ET include availability of water, energy supply to water, soil, vegetation, type of surface, wind, atmospheric pressure, and temperature (CCL, 2001). Several methodologies for estimating PET and AET exist. The methodologies vary in complexity, input requirements and source of derivation. The methodologies for estimating PET can be grouped into several categories including empirical, mass transfer, combination, radiation, temperature, and measurements (Oudin et al., 2005). Oudin et al. (2005) conducted a study reviewing 27 PET models to determine how atmospheric variables can be used to estimate evaporative demands at a basin scale. The same study found that several less data intensive PET methods were nearly as effective in daily rainfall-runoff modelling as more data intensive PET methods.

The more commonly utilized (CCL, 2001, Oudin et al., 2005, and Ritter, 2006) PET calculation methods include Penman-Monteith (Monteith, 1965), Priestley-Taylor (Priestly and Taylor, 1972), Thornthwaite (Thornthwaite, 1948), and Turc (Xu and Singh, 2001). The methodology used to calculate PET in wetland WBs is as variable as the methodologies for determining PET. It is also common to estimate AET using the pan evaporation method normally corrected by an adjustment factor (Gehrels and Mulamoottil, 1990; Mitsch and Reeder, 1992; CCL, 2001).

Soucha et al. (1996) conducted a 10 day study in a fresh water coastal wetland, the Great Marsh, outletting to Lake Michigan. In the same study, ET was measured directly as its energy equivalent, the latent heat flux, using eddy correlation techniques. Several models for determining ET were compared to the values calculated using the recorded heat fluxes. In the Soucha et al. study (1996) it was shown that the Penman and Priestley-Taylor ET models were successful at predicting water exchange rates in the Great Marsh wetland. The hysteresis data intensive model was found to be much better at predicting hourly rates, but on larger time scale (i.e. daily), the Penman and Priestley-Taylor were found to be appropriate.

In permanently flooded wetlands an unlimited supply of water is available which would suggest that AET would be PET in this scenario. A study by Shoemaker and Sumner (2006) studying the methods of correcting PET rates to AET rates in wetlands found that this was not necessarily the case. The study considered humid subtropical wetlands dominated by open water, short saw grass, rushes, and cattail vegetation; in specific, nine wetland Everglade sites located in the southern region of Florida, USA. The investigation (Shoemaker and Sumner, 2006) found that previous studies have found that AET rates were less than PET rates, even when an unlimited source of water was available.

AET is a complex component of the WB to analyze and quantify. Favero et al. (2007) investigated methods of estimating different components of a wetlands WB, with the intent of determining the best way to develop a WB useful for application and design purposes of artificial or partial artificial wetlands. AET was estimated using the Penman Monteith equation, and seepage, was estimated using Darcy's law. It was found that the highest uncertainty in the WB was in the seepage and ET components.

The components of a WB are all linked as outlined in Equation 2.1. Smithers et al. (1995) investigated the potential of using daily groundwater table fluctuations and soil retention properties to estimate missing components in a WB, in specific the potential for estimating AET was considered. The study area examined the hydrologic balance of the Ntabamhlope freshwater riparian wetland in the eastern region of South Africa. To determine ET losses with this method, it is assumed lateral groundwater flow is constant and any fluctuations in groundwater flow are caused by ET. This method is relatively simple to perform, and inexpensive. Days with rainfall cannot be used to calculate ET: this drawback is also exemplified in an over estimation of ET rates, as ET cannot be estimated on wet days.

2.3.2 Wetland Surface Flow Processes

Surface water processes in wetlands include streamflow, flooding from lakes and rivers, overland flow, and tidal flow (WRP, 1993a). In general, surface water outflow from wetlands is greatest during the wet season and wetlands that have a large component of groundwater inflow tend to have more evenly distributed outflow streamflow throughout the year (USGS, 1999). The relative importance of surface water processes varies with each unique wetland depending on climatic, geological, and geographic factors.

Despite the variability of each surface water flow effect on a wetland, wetlands affect surface water process and wetlands are affected by surface water processes. Wu and Johnston (2008) investigated the hydrologic relationship between heavily forested watersheds and watersheds with large open bodies of water (wetlands and lakes). The Soil and Water Assessment Tool (SWAT) model was utilized to simulate the hydrology of two northern Michigan watersheds. Wu and Johnston (2008) study confirmed that snowmelt was a major contributor to runoff and watershed chemistry with approximately half of the annual precipitation occurring as snowfall. The watershed with the higher proportion of wetland/lake cover (26 % compared to 17 %), showed a reduction in maximum flow rate during the spring melt, but a sustained higher flow in the summer time. These results suggest that wetlands serve an important function in regulating and reducing peak flows in watersheds. The two watersheds had a similar average annual flow rate, but the monthly averages varied significantly due to the wetland storage capacity.

In coastal wetlands the adjacent water body and the barrier play a fundamental role in the hydrology of the region. Mitsch and Reeder (1992) conducted a nutrient and hydrologic budget in a fresh water coastal wetland at the Old Woman Creek State Nature Preserve and National Estuarine Research Reserve adjacent to Lake Erie in Erie County, Ohio. The components accounted for in this WB were streamflow entering the marsh from the watershed, direct precipitation, ET, and flow entering or leaving the marsh from Lake Erie. It was found that one of the most significant (and rapid) components of the WB was flow through the disturbed beach barrier. This component could represent both flows into and out of the wetland.

Wetlands are complex systems where the dominate flows will vary based upon ambient conditions. In a study by McKillop et al. (1999), highlighting the importance of external factors on wetlands, a hydraulic model was constructed and reviewed to emulate the hydrology of a wetland in south-western Ontario, Canada. The study wetland, approximately 400 hectares in area, composes the headwater for a larger drainage basin. The model emulated both surface and subsurface flow. The model consists of both a channel routing system which is connected to a hydrologic wetland model. The wetland was discretized into a grid network (with grid blocks ranging from .6 to 33 m). PET was calculated with the Turc equation. A sensitivity analysis was conducted on all model input parameters, where each parameter was altered by five percent. In the sensitivity analysis, precipitation was found to be the most sensitive variable. The model was also found to be sensitive to ground saturation conditions. With low saturation the modeled streamflows were found to be governed by near stream subsurface properties and the sensitivity to slope was found to be minimal. Conversely, under highly saturated conditions overland flow governed and parameters related to surface flow were found to be more sensitive.

Wetlands associated with lakes and streams store floodwaters by providing temporary storage of water decreasing runoff velocity, and reducing peak flows (USGS, 1999). Owen (1995) conducted a WB study on an urban wetland in Monona, Wisconsin, USA. The study wetland had a peat soil bed, and an area of 92 hectares. The major inputs for the WB were streamflow and precipitation. The study found that the wetland did not significantly contribute to groundwater recharge which accounted for less than one percent of the WB's total outflows. The wetland has a large storage capacity and could be used for flood control during storms.

2.3.3 Wetland Subsurface Flow Processes

Subsurface flows, like surface flows in wetlands, vary as greatly as the characteristics of individual wetlands (USGS, 1999). Groundwater processes can be divided into two flows, shallow and deep, and are primarily influenced by four major factors; hydraulic gradients, hydraulic conductivity, porosity, and storage coefficient (WRP, 1993b). Shallow and deep ground water zones can be independent (separated by an impervious layer), or be coupled and exchange groundwater. A hydraulic gradient is the difference in piezometric head at two locations divided by the distance (Bedient et al., 2008). The hydraulic conductivity is the ability of a soil to flow water under hydraulic gradients.

Three in-situ methods of groundwater measurements were compared in a riparian wetland in the Hunt et al. (1996) study. The research work highlighted the complexity of groundwater processes in wetlands. The study site was a natural and constructed wetland along the Kickapoo River in Monroe County, Wisconsin. The difficulties in quantifying groundwater flow included aquifer heterogeneities, complex properties of peat, and seasonal variation in hydraulic gradients. The three groundwater measurement techniques studied in Hunt et al. (1996) were stable isotope mass balance, temperature profile modelling, and numerical water balance modelling techniques. In addition, a simple calculation using Darcy's law was included. This alternative comparison was included because wetland WBs frequently estimate groundwater flow using Darcy's law. It was found that both the numerical modelling method and the simple Darcy law calculation produced lower estimates of groundwater flow. The use of Darcy's law yielded the smallest groundwater flow, and the pattern of increasing flow rate proportional to river proximity was also not modelled with Darcy's law, as was found in the other three methods.

The subsurface components of a wetland's WB require significant data to model. Favero et al. (2007) conducted a wetland WB study and found that the highest uncertainty was related to seepage and ET. The study estimated seepage using Darcy's law with measurements of the site's piezometric heads and hydraulic conductivity. The piezometric heads were initially measured every 18 days in the study, but a more frequent measure of head, five day period, reduced the WB error (from 25 to 13 %). The complexity of estimating groundwater processes in wetlands is linked with the great variability of subsurface properties. The spatial variability of soil properties often requires an uneconomical number of soil surveys to model hydraulic conductivity of a large site, as hydraulic conductivity measurements can vary by orders of magnitude, even in a relatively homogeneous aquifer (Favero et al., 2007).

Climatic conditions have dramatic influence over the WBs in wetlands. Frozen conditions can alter soil's hydraulic conductivity. Hayashi and Van Der Kamp (2009) reviewed field studies related to groundwater recharge in North American prairie wetlands. They found during early periods of snowmelt, frozen soils allow for surface flow to be the significant input to prairie wetland WBs. Undisturbed, natural soils have a substantial micropore density giving even the frozen soils high hydraulic conductivities. It was found that disturbing the top layer of soil will reduce the soil's hydraulic conductivity and drastically change flow regimes, reducing wetlands total inflow. Finally it was concluded shallow groundwater lateral flow was a governing component of inputs and outputs between wetland ponds and the riparian zone (area of dense vegetation surrounding ponded wetland water). The Hayashi and Van Der Kamp (2009) study shows the great variability of wetland WBs being sensitive to climatic settings and anthropogenic upstream effects.

2.4 Wetland Water Budget Modelling

The movement of water in a wetland significantly influences wetland functions and characteristics; consequently the hydrology is of primary importance in evaluating wetlands (WRP, 1997). To better understand how wetlands function and respond to hydrological, meteorological, and landuse changes modelling is necessary (Erwin, 2009), especially in the case of reviewing the effect of climate change. Wetland modelling is fundamental for planning, management, and restoration in these ecosystems. A wetland WB model would follow the same mass balance equation as any other WB model, but the unique hydrologic, geographic, and ecologic features would need to be incorporated.

2.4.1 WDWBM

The Wetlands Dynamic Water Budget Model (WDWBM) was developed by the WRP to predict the interaction of surface water, groundwater, and vertical transport processes of water within wetlands (WRP, 1997). The model is a coupled surface/aquifer simulation program that computes the dynamic movement of water in wetlands. The dynamic movement of water is approximated with three different modules; surface water flow, vertical processes, and horizontal groundwater flow. The WDWBM was calibrated and validated on the Black Swamp portion of the Cache River using data from 1988 to 1991 (Walton et al., 1997). The WDWBM model was found to accurately simulate in-bank water levels, overbank water levels, and downstream flows but additional research is still required for model development and verification (Walton et al., 1997).

2.4.2 WG-WETLAND

The WG-WETLAND (Shikaze and Crowe, 1999) model can be used to simulate transient groundwater flow and contaminate transport in a variety of groundwater-wetland environments. The model simulates two dimensional groundwater flow, particle tracking, and solute transport with transient boundary conditions and a fluctuating water table. The WG-WETLAND model was applied to the Point Pelee wetland in Ontario, Canada (Crowe et al., 2004). The results of the same study were in agreement with field observations following the same seasonal reversal of flow as was observed onsite. To both model with and verify the results of the WG-WETLAND model, quality input data needs to be provided by the user (Crowe et al., 2004).

2.4.3 FEUWAnet

The FEUWAnet model simulates riparian wetland water systems exchanging water using a series of linked boxes (Dall'O et al., 2001). The boxes emulate wetlands hydrologic processes by acting as storage (open water, soil storage) linked to other boxes by hydrologic resistances, composing a series of differential equations. The model allows for spatial and temporal variation of lateral and vertical flows. The model was used in a study (Dall'O et al., 2001) to simulate a riparian wetland adjacent to Lake Belau in northern Germany. The modelling results of the FEUWnet study (Dall'O et al., 2001) show a good fit to both the observed water levels and an independently calculated WB.

2.4.4 REMM

The Riparian Ecosystem Management Model (REMM) simulates riparian hydrology and nutrient processes zones near a stream; however, it requires upland inputs from either a separate model or from observed field data (Arnold, et al., 2009). The model was developed by the US Department of Agriculture, Agricultural Research Service to predict the effect of NPS pollutant reduction in riparian vegetative zones by emulating surface and subsurface flow, nutrient cycle, sediment transport, and vegetative growth (Ik-Jae et al., 2007). The hydrologic components of the REMM are governed by both mass balance and rate controlled approach processes. The subsurface processes simulated in the model include vertical drainage based upon soil layer field capacity and lateral flow calculated using Darcy's equation (Langendoen and Lowrance, 2009).The REMM model simulates individual hillsides or fields with a daily time step. The model has been evaluated and validated to accurately predict hydrology, water quality and nutrient cycling on the field scale (Ik-Jae et al., 2007).

In the Ik-Jae et al. (2007) study, a sensitivity analysis was preformed with the REMM. In the same study it was found that altering Manning's n coefficient (by 50 percent) had a robust effect on surface flow transport processes, significantly increasing sediment yield and total nitrogen yield. The REMM was not found to be sensitive to changes in vegetation model inputs. The model was sensitive to metrological inputs suggesting that onsite weather data be used in modelling.

In the Langendoen and Lowrance study (2009) the REMM was evaluated on its ability to simulate seepage and soil water distributions within a wetland. The model was tested against data collected from a lysimeter experiment and two alternative subsurface flow models. The REMM adequately predicted pore water pressure distribution on daily and larger time scales, but greatly under-predicted seepage outflow. It was concluded from the study that REMM should not be used to calculate seepage-induced erosion in riparian zones.

2.4.5 VIC

Mishra et al. (2010) conducted a study using the Variable Infiltration Capacity (VIC) model to emulate the hydrological interaction between wetlands and lakes. The VIC model is a large scale, semi-distributed hydrologic model. Model inputs include climate data, necessary to simulate land-atmosphere water and energy fluxes, and streamflow, which is routed with an independent model. One of several model inputs included historical weather data from the fresh water coastal (Great Lake) states of Michigan, Wisconsin, and Minnesota. Several watersheds were considered in the study region, and data permitting the model was calibrated from 1985-1995 and the model was validated from 1996-2005. The VIC model does not model on the wetland scale, but on a much larger scale. The model simulated the hydrology of the study area and found that in simulations without lakes and wetlands, ET would decrease by 28 mm (5 %).

2.4.6 Other Wetland Water Budget Models

McKillop et al. (1999) constructed a hydraulic model to emulate the hydrology of a wetland. The McKillop et al. (1999) study shows the effectiveness of a depth-average laminar model coupled to a stream-routing model suitably to simulate both "short-duration and long-duration flows from headwater wetland environments." The model (McKillop et al., 1999) consists of a channel routing system which is connected to a hydrologic wetland model. The study wetland was converted into a grid network (with grid blocks ranging from .6 to 33 m). The model requires an independent input for groundwater flow. McKillop et al. (1999) concluded that the model reproduced reasonably accurate rainfall-runoff results, but caution should be considered when assuming the validity of such a complex model with the use of streamflow being utilized as the only evaluation criteria (Nash-Sutcliffe coefficient ranging from - 0.10 to 0.95 for single event simulations). A sensitivity analysis was conducted on all model input parameters, where each parameter was altered by five percent. In the sensitivity analysis, precipitation was found to be the most sensitive variable. Additionally parameters related to wetland organics were also deemed to be significant including porosity, hydraulic conductivity, and organic layer thickness.

Krasnostein and Oldham (2004) developed a conceptual model to emulate the different hydrologic components affecting a wetland. A bucket storage model was implemented in the simulation of the permanently flooded Loch McNess wetland located in Perth, Western Australia. The bucket storage model used leaky buckets in series and/or parallel to emulate a wetlands hydrologic inflows and outflows. Elements in the WB measured include flow from the wetlands catchment, groundwater flow, and outlet lake flow. The linked buckets had a saturated capacity (depth of water), seepage, and overflow functions. The study (Krasnostein and Oldham, 2004) results showed that the bucket model could be used to emulate the hydrological processes of a wetland system. The conceptual bucket model was only tested in a single wetland, but could be applied to a boarder range of wetlands (Krasnostein and Oldham, 2004).

2.4.7 Wetland Chemical Balance Modelling

Wetland WBs serve as the basis to better understanding the chemical and ecological processes within the same ecosystem. Both Mitsch and Reeder (1992) and Gehrels and Mulamoottil (1990) conducted a nutrient and hydrologic budget in fresh water wetlands. Mitsch and Reeder's (1992) research work consisted of a simple WB and phosphorus mass balance in the barrier beach wetland. The nutrient budget looked specifically at phosphorus which was calculated using only hydrology and water chemistry. Phosphorus cycling by plankton was calculated from metabolism measurements, with an assumed uptake rate. It was determined (Mitsch and Reeder, 1992) that phosphorus was being deposited in the ecosystem over the study year. However, only one year was analyzed in the study therefore only limited conclusions should be drawn regarding the wetlands functionality as permanent sink for phosphorus.

Gehrels and Mulamoottil (1990) conducted a twelve month hydrologic study in a wetland, Hidden Valley, located in Kitchener, Ontario. The subject wetland is a Typha marsh 18 hectares in size, bordered by wooded esker, hardwood forest and non-wooded agricultural landuse. During the study period both the hydrologic processes and the phosphorus balance of the wetland were analyzed. Both the mass flux of water and concentrations of phosphates, ortho-phosphates, and chlorides were measured in all components of the WB. The phosphate input from precipitation was determined by multiplying the amount of precipitation for each season by median concentrations. Surface water samples were taken twice weekly and as necessary during storm events. Groundwater flow was estimated using ten bi-level piezometers (both at 1 m and 3 m depths). Using the piezometers hydraulic gradient readings were taken weekly and groundwater samples were taken biweekly.

2.5 Non-point Source Pollution

Non-point source (NPS) pollution can simply be defined as pollution that is generated over an area that cannot be tracked to a single point. The term NPS is used to distinguish it from point source pollution, which comes from localized, easily identifiable sources such as sewage treatment plants or industrial facilities (Russell and Shogren, 1993). Surface water runoff collects natural and man made pollutants then transports the contaminates to rivers, streams, wetlands, and other bodies of water (Subra and Waters, 1996). Once the pollutants enter the water body the environmental and ecological impact can vary substantially. Common NPS pollutants of environmental concern include sediment, nutrients, acids and salts, heavy metals, and pathogens (Leeds et al., 1992).

In the Great Lakes Basin most point sources of toxic loadings are well understood and controlled, however the biggest remaining problem with controlling water quality in the basin is NPS pollutants (USEPA, 1997). The environmental concern with NPS pollutants is not the material itself, but its concentration; nutrients such as nitrogen and phosphorus are essential elements for plant growth. If they are overabundant in a body of water, this can lead to conditions that have a negative effect on people's health (NOAA, 2007).

There are several benefits that can result from the tracking, monitoring, simulating, and controlling NPS pollutants including improvement in water quality, ecological rehabilitation, and recreational benefits. Modelling NPS pollutant fate and transport processes across multiple scales is fundamental to addressing several environmental and natural resource issues, including the degradation of soil and contamination of surface and ground waters (Srivastava et al., 2007). Before launching any new activities within a watershed or incorporating any remedial measures, it is vital for decision makers and planners to evaluate the consequences of these actions (Oogathoo, 2006).

2.5.1 Watershed and Non-Point Source Models

NPS pollution is generated over a large geographical region and is transported by the drainage of water through a watershed; consequently, NPS pollution modelling is commonly an extension of hydrologic watershed modelling tools. In the most general sense watershed models can be categorized with two different scales, temporal and spatial. From the temporal scale, watershed models can be continuous or single-event. From the spatial scale, watershed models can be lumped parameter or distributed parameter models (Jayasuriya, 2007). A lumped modelling approach considers a watershed as a single unit, using the spatial averages for model calculations, whereas distributed models account for the spatial variability of hydrologic processes, input, boundary conditions, and watershed characteristics (Daniel et al, 2011).

Hydrologic models are commonly used to simulate NPS pollution. Four of the more common continuous distributed parameter watershed NPS models include AnnAGNPS, SWAT, ANSWERS-2000, and HSPF (Jayasuriya, 2007 and Rahman, 2007). The four above models will be briefly reviewed.

2.5.2 AnnAGNPS

The AnnAGNPS (Annualized AGricultural Non-Point Source) model is a continuous time simulation model that can assess the impacts of landuse management strategies. The model is able to assess landuse alternatives because it can track both point source and NPS pollutant loadings that are produced within a watershed. The AnnAGNPS model uses an ArcView Geographic Information System (GIS) interface to simplify the modelling process. The pollutant loadings that can be modeled with AnnAGNPS include pesticides, nutrients and sediments. AnnAGNPS was originally designed to track pollutant loadings in agricultural watersheds (Gordon et al., 2007).

In 1998 the first version of the AnnAGNPS model was released. The AnnAGNPS modelled watersheds are divided into homogeneous land areas based on landuse, soil type and land management (Bingner et al., 2009). These land areas, called cells, are the origin of NPS pollutants in the model, which are transported through the stream network, and may be destined for the watershed outlet. In AnnAGNPS peak flow calculations are performed using TR-55 graphical peak discharge method (NRCS, 1986). Runoff volume is calculated using the SCS Runoff Curve Number method, where the curve numbers are modified daily, based upon soil moisture, crop stages and tillage operations. A daily mass balance for nitrogen, phosphorous, and organic carbon are calculated for each cell. Both nutrients and pesticides are subdivided into soluble and sediment attached components for routing. Each nutrient component is decayed based upon the reach travel time, water temperature, and an appropriate decay constant (Bingner et al., 2009).

The AnnAGNPS model uses Simultaneous Heat and Water Transfer Model to account for snowpack melt, snowpack accumulation, snowpack compaction, soil/snow temperature profile, soil/snow moisture profile and latent heat transfer processes (Moore et al., 2006). The variables most influential variables for snowpack and snow melt are meteorological; maximum daily temperature, minimum daily temperature, dew point temperature (Moore et al., 2006). The AnnAGNPS model is described in greater detail in Chapter 3.

2.5.3 SWAT

Soil and Water Assessment Tool (SWAT) is a continuous, physically based, distributed parameter watershed scale model developed to predict the impact of land management practices on water, sediment, and chemical yields in complex watersheds that was developed for the Agricultural Research Services of US Department of Agriculture (Neitsch et al., 2005). SWAT delineates the modelled watershed into several smaller subwatersheds called Hydrologic Response Units (HRUs). An HRU is a lumped land area that has unique land cover, soil and land management conditions throughout the unit. The SWAT model was developed from the SWRRB (Simulator for Water Resources in Rural Basins) model and is the culmination of nearly thirty years of modelling efforts (Arnold, 1987 and Gassman et al., 2007). The SWAT model inherits features from several other models including the CREAMS (Chemicals, Runoff and Erosion from Agricultural Management Systems), the GLEAMS (Groundwater Loading Effects on Agricultural Management Systems), and the EPIC (Erosion – Productivity Impact Calculator) models (Neitsch et al., 2005).

The SWAT model uses a GIS interface to enable automatic development of spatially varied inputted model parameters. The SWAT model's hydrology components have been validated in several watersheds (Arnold, et al., 2009). The hydrologic balance for each HRU includes the WB components of canopy interception, partitioning of precipitation, snowmelt, irrigation, ET, lateral subsurface flow, and return flow from shallow aquifers (Gassman et al., 2007). Sediment yield in the model is calculated with MUSLE (Modified Universal Soil Loss Equation), mass balances of nutrients are calculated in each HRU using the supply and demand approach, and PET is estimated using one of three equations: Hargreaves, Priestley-Taylor, and Penman-Monteith (Neitsch et al., 2005).

2.5.4 ANSWERS-2000

The ANSWERS (Areal Non-point Source Watershed Environment Response Simulation) -2000 model is also a continuous simulation, distributed parameter, physically-based model (Dillaha et al., 2001). The model development began in the late 1970s (entitled ANSWERS) as an event-oriented planning model to evaluate management practices, sediment and runoff effects on agricultural watersheds; in the late 1980s phosphorous and nitrogen transport systems were added to the model (Dillaha et al., 2001). The event based version of the model ANSWERS has been successfully validated for NPS pollution evaluation (Bouraoui and Dillaha, 2000). The model represents a watershed as a matrix of uniform square elements, where all parameters in each element are considered uniform, maximum element size is one hectare, and parameter values in an element are unrestricted, allowing for any degree of spatial variation between individual elements (Bouraoui and Dillaha, 1996).

The continuous version, ANSWERS-2000, was evaluated on two watersheds in Watkinsville, Georgia and a third watershed in Virginia (Dillaha et al., 2001). It was concluded from the same study that the model predicted runoff, sediment, and certain nutrients from the watershed well. In the Parson et al. (2001) review of the ANSWERS-2000 model, several limitations were highlighted. These areas of model improvement included; replacing the existing empirical sediment detachment model with a physically based model; updating the nitrogen processes in the model, with specific concern for urban areas; adding a channel erosion and scour process subroutine. The final noted limitation was related to incorporating subroutines to emulate the effects of buffers, wetlands, and detention ponds.

2.5.5 HSPF

The Hydrological Simulation Program - FORTRAN (HSPF), developed by USEPA is a lumped parameter, continuous simulation model that can simulate watershed hydrology and water quality for pollutants (Bicknell, 1996). HSPF models all streamflow components including surface runoff, interflow, baseflow, and their pollutant contributions. The HSPF model's first incarnation was developed in the early 1960's as the Stanford Watershed Model. The model was continuously updated though the decades to include water quality processes and software upgrades (USGS, 2010).

The HSPF model has been widely used in the United States and around the world for both research and engineering proposes, and has been validated numerous times (Bai, 2010). The model has been successfully applied to a variety of watershed locations, sizes, and study purposes. The study by Borah and Bera (2004) conducted a review of NPS pollutant loading models. In the same review (Borah and Bera, 2004) twelve studies with the HSPF model involving model calibration, model validation, landuse practice management, and other purposes were outlined. In the review a noted common drawback of HSPF, as with many other hydrologic models, was the numerous data requirements of the model, thus making it difficult to procure the data needed for modelling and making the calibration process more complex.

2.6 AnnAGNPS Studies

The following section provides a literature review of past AnnAGNPS model studies. The review focuses on AnnAGNPS calibration and validation studies, AnnAGNPS studies in Ontario and the Great Lakes Basin, and AnnAGNPS studies reviewing other portions of the model.

2.6.1 AnnAGNPS Calibration and Validation

The following section contains a review of AnnAGNPS calibration and validation studies. The output parameters of the AnnAGNPS model that were validated in the reviewed studies include surface runoff, nutrient yields, and sediment yields. The process of model calibration evolves estimating model input parameters through their adjustment to match a modelled output with observed data (Rahman, 2007). Model calibration is an iterative process of modifying the model inputs until the model outputs have a minimized average error or relative error. The validation process assesses the modified input parameters under alternative conditions and the models predicted output to the observed data.

Yuan et al. (2006) investigated the subsurface flow component of the AnnAGNPS model by studying the Upper Auglaize Watershed in Ohio, USA. The model was only calibrated with annual average runoff data because the historic metrological data for the region was not available, thus synthetic weather data was used for modelling. One hundred years of synthetic weather data were utilized in the AnnAGNPS model. The model was calibrated with both 24 years of annual average sediment and flow data. The model's predicted annual average sediment loading and runoff were nearly equivalent to the observed data.

In Parajuli et al. (2008) a comparison of the SWAT and the AnnAGNPS models was investigated. Each model was validated and calibrated with three distinct metrics; surface runoff, sediment yield, and total phosphorous. The research study focused on two neighbouring watersheds in south-central Kansas, one watershed for calibration and the other for validation. Both models performed fair to very good with the monthly flow in the calibration and validation watersheds (considering the Nash-Sutcliffe efficiency, correlation coefficient, and root mean square error). For flow calibration only curve numbers (CNs) were adjusted.

In the same study (Parajuli et al., 2008), sediment calibration was found to be a function of the universal soil loss equation's C-factor USLE for both SWAT and AnnAGNPS. The calibration and validation of monthly sediment yield for each model was good (considering the same indices as streamflow). The two models were not calibrated for total phosphorus loading but were evaluated in both the calibration and validation periods. Both models had a good to fair agreement in the calibration watershed; however, AnnAGNPS significantly over-predicted the monthly phosphorous loadings in the validation watershed. The over-prediction of total phosphorous was noted as a common issue with AnnAGNPS in the literature review by Parajuli et al. (2008).

Kliment et al. (2008) also compared calibrated SWAT and AnnAGNPS models studying both sediment yielding and streamflow. The study watershed for the research was the Blšanka river basin in the Czech Republic. The study watershed was calibrated with five years of data and validated with an alternative five years. In the calibration for this study adjusted input parameters of the model were hydraulic conductivity, field capacity, and CN. In the study it was found that AnnAGNPS under-predicted total streamflow by over 50 percent in both the validation and calibration periods. The SWAT model generally performed better for predicting annual streamflow. The daily streamflow analysis for both the Nash-Sutcliffe efficiency and the coefficient of correlation were poor for SWAT and AnnAGNPS in both calibration and validation, this is likely because the daily values were used for analysis. Similar results were predicted for suspended sediment loadings; however, the sediment values for both models matched the observed data better for total magnitude, with the AnnAGNPS predictions matching slightly better.

Pease et al. (2010) investigated the applicability of the AnnAGNPS model in the eastern central North Dakota region. The watershed considered in this study is approximately 1 697 km^2 in area. A 30 m by 30 m DEM was utilized to determine topographic parameters. The model was calibrated with observed streamflow data. The only parameter used for model calibration was the CN, being adjusted for all landuse types. The model was not validated in the study. Both the nutrient and sediment loading predicted by the model did not match well with the observed data: of the 23 events observed during the study period only 4 had non-zero values in AnnAGNPS. The large land size is considered a detriment to the modelling, as the area was suggested to be a cause of the poor correlation between the observed and modelled nutrient and sediment data.

In the Yuan et al. (2005) study the AnnAGNPS model was validated and calibrated in the Deep Hallow Watershed in Leflore County, Mississippi. The intent of the same study was to investigate the phosphorus modelling components in AnnAGNPS: thus in this study the model was only calibrated and validated with respect to the soil's initial organic and inorganic phosphorous content. The other AnnAGNPS variables were inputted from the model's included database for the study's geographical location.

The results of the study (Yuan et al., 2005) found that AnnAGNPS can be successfully calibrated and validated for total monthly phosphorus loading, producing both good Nash-Sutcliffe and coefficient of correlation results for both the validation and calibration phases. Other AnnAGNPS studies (Pease et al., 2010 and Parajuli et al., 2008) found weaker matches between predicted and observed nutrient loading; this inconsistency could be easily accounted for by not properly estimating nutrient model inputs.

2.6.2 AnnAGNPS Studies in the Great Lakes Basin

Gebremeskel et al. (2005) assessed the performance of several models applicability to simulate streamflow and sediment processes in Ontario conditions. AnnAGNPS was one of several models considered in the study, including HSPF, SWAT and ANWSERS-2000. All models in the study were calibrated and achieved a good Nash-Sutcliffe coefficient. The study found that AnnAGNPS under-predicted runoff and over-predicted the sediment yield, but suggested that the model had the capability of simulating runoff and sediment yield fairly well for a cold and temperate region like Ontario (Gebremeskel et al., 2005).

Jayasuriya (2007) investigated the applicability of the AnnAGNPS model in the Essex County region of Ontario. The Muddy Creek Watershed, an ungauged watershed, was investigated in the research. The predicted annual average runoff was compared to neighbouring gauged watersheds, but was found to be slightly lower. Sediment loading and nitrogen loading were comparable to other watersheds in literature. AnnAGNPS was found to perform reasonably well, as a hydrological NPS model, given the limited comparable statistics for model verification. In the same study (Jayasuriya, 2007) it was concluded that an area of further research could include model calibration and validation with continuous streamflow, sediment, and/or nutrient data.

Two studies (Das et al., 2007 and Das et al., 2008) of the Canagagigue Creek Watershed investigated AnnAGNPS validity and functionality in Ontario, Canada. In the first study (Das et al., 2007) a comparison of the SWAT and the AnnAGNPS models was undertaken. Both models were calibrated and validated using five years of data for each phase. The calibration considered annual and monthly surface runoff. In general, SWAT over-predicted runoff. AnnAGNPS, however, over-predicted runoff from June to December, and under-predicted in the other months. Das et al. (2007) suggested that the over and under-predicting of AnnAGNPS could be caused by AnnAGNPS not being able to address frozen conditions, semi-frozen conditions, and snowmelt. In the same study AnnAGNPS had better monthly direct runoff Nash-Sutcliffe coefficients than the SWAT model. The AnnAGNPS predictions produced very good and good Nash-Sutcliffe coefficients in the calibration and validation periods, respectively. The AnnAGNPS performed fairly well in simulating runoff and sediment processes in Ontario conditions, with room for improvement in modelling late winter and spring processes (Das et. al., 2007).

In the second study (Das et al., 2008) of the Canagagigue Creek Watershed a greater detailed investigation of the AnnAGNPS model in Ontario conditions is outlined. The predicted daily runoff generally matches the timing of the observed runoff peaks. In general, AnnAGNPS over-predicted sediment loading with a Nash-Sutcliffe coefficient of good for the calibration and poor for the validation phase. The conclusion in Das et al. (2008) matched the conclusions from the older study (Das et al., 2007), confirming that AnnAGNPS performs reasonably well in emulating runoff and sediment in Ontario.

2.6.3 AnnAGNPS Studies Investigating Other Model Components

In the Yuan et al. (2011) study the AnnAGNPS modelled is evaluated on the effect of delineation size of the cells composing the watershed. The study watershed was in East Fork Kaskaskia River Watershed in Illinois, USA. The model was calibrated with the available streamflow data but was not validated. The study found that all delineations of the watershed produced satisfactory runoff results, but the benefits of finer delineations were in the better representation of the actual landscape to target areas of concern (Yuan et. al, 2011). The monthly runoff had a good and excellent Nash-Sutcliffe coefficient and coefficient of determination, respectively. The annual average nitrogen loading predicted by the model was less than the observed data. The variations of cell delineations sizes did not affect the annual average nitrogen loading rates, but refined the source of the NPS nutrients.

In Jincheng et al. (2010) study an investigation of the applicability of the AnnAGNPS model in the Karst area of Guilin, China was conducted. The study was calibrated with annual average total nitrogen and phosphorus loading at the watershed outlet; however, the calibrated model was not validated in the same study. Nutrient loading was found to be highly dependent on sediment transport; as both phosphorus and nitrogen are frequently bound to sediment particles. The difference between the simulated and observed total nitrogen and phosphorous were 11.5 and 23.0 %, respectively. Without validation results from this study limited conclusions should be drawn. AnnAGNPS was able to simulate annual average nutrient loadings over the study period.

Jayasuriya (2007) investigated the AnnAGNPS model's applicability in the Muddy Creek Watershed in Essex County, Ontario, Canada. The study's modelling results were un-calibrated, but included a substantial sensitivity analysis of the AnnAGNPS model. The effect of cell size variation was investigated on multiple model outputs. The sediment loading at the watershed outlet was found to be the most sensitive to these changes with a cell increase from 1.0 ha to 20.0 ha, a reduction in sediment loading of approximately 66 times occurred (Jayasuriya, 2007). It was suggested that grid cell sizes in the delineation should be selected such that the flow path lengths in the model approximate the actual drainage network. A summary of the AnnAGNPS sensitivity analysis by Jayasuriya (2007) is outlined in Table 2.1. Das et al. (2008) had confirmed that the same variables were sensitive for runoff and sediment yield.

Table 2-1: AnnAGNPS Model Sensitivity

Model Output	Sensitive Model Inputs
Runoff	wilting point, field capacity, CN
Sediment Yield	K factor, wilting point, field capacity, surface roughness
Nitrogen Loading	wilting point, field capacity, plant nitrogen uptake, initial organic nitrogen in soil, initial inorganic nitrogen in soil, CN
Phosphorous Loading	K factor, wilting point, field capacity, plant phosphorus uptake, initial soil organic phosphorus, initial soil inorganic phosphorus, CN

(source: Jayasuriya, 2007)

2.7 Summary

This chapter provided a short review on the general theory of WBs. An extensive review of wetland WB and WB modelling was outlined. Wetland geographical, geological, and climatic properties vary substantially from site to site: consequently so does the significance of each WB component from wetland to wetland. The wetland WB models reviewed vary in complexity, function, and model validation. The fundamental mass balance of hydrologic inflows, outflows and storage was present in all reviewed literature. The most complicated aspects of wetland WB modelling are approximating estimates of ET and subsurface water processes. An extensive review found that there wasn't a commonly used wetland WB model that has been validated and calibrated as extensively as the watershed models reviewed in the second half of the chapter.

The chapter also provided a review of NPS loading and NPS loading models with an emphasis on studies pertaining to AnnAGNPS. The NPS models reviewed were AnnAGNPS, SWAT, ANSWER-2000, and HSPF. The AnnAGNPS model has been validated in a variety of geographical regions and conditions. In particular, AnnAGNPS was found (Gebremeskel et al., 2005, Jayasuriya, 2007, Das et al, 2007, and Das et al., 2008) to be a reasonable NPS and watershed model in Ontario conditions.

CHAPTER 3: AnnAGNPS MODEL REVIEW

3.1 Introduction

The following chapter provides a comprehensive review of the AnnAGNPS (Annualized Agricultural Non-Point Source) computer simulation model which was utilized to emulate the Big Creek Watershed's hydrological, chemical, and physical processes for a twenty year period from 1990 to 2009. This chapter will provide a brief albeit in-depth review of the model's historic development, input methodology, and the equations used to simulate the watershed's non-point source loadings. The majority of the model review below is taken from the AnnAGNPS Technical Processes (Bingner et al., 2009).

3.2 AnnAGNPS MODEL OVERVIEW

The AnnAGNPS model is a continuous time simulation model that can assess the impacts of land use management strategies. The model is able to assess landuse alternatives because it can track both point and non-point source pollutant loadings on a watershed scale. Non-point source pollution is generated over a large plot of land, whereas point source pollution is traceable to a single location. The pollutant loadings that can be modeled with AnnAGNPS include pesticides, nutrients and sediments. AnnAGNPS was originally designed to track pollutant loadings in agricultural watersheds.

The first version of the AnnAGNPS model was released in 1998 to evaluate non-point source pollution in agricultural watersheds up to 3000 km^2 in size. In AnnAGNPS, watersheds are divided into homogeneous land areas based on land use, soil type and land management (Bingner et al., 2009). These land areas, called cells, are the origin of non-point source pollutants in the model, which are transported though the stream network, and may be destined for the watershed outlet.

The AnnAGNPS model uses meteorological, hydrological, and other physical processes to determine pollutant loading. AnnAGNPS is a modification of a single event model AGNPS developed in the early 1980's by the Agricultural Research Service (ARS) and Natural Resources Conservation Authority (NRCS) (both NRCS and ARS are departments of the USDA, United States Department of Agriculture).

AnnAGNPS calculations are performed on a daily time step. Each day the applied water and resulting runoff are routed through the watershed system before the next day is considered. Runoff volume is calculated using SCS Runoff Curve Number equation where the curve numbers (CN) are modified daily, based upon soil moisture, crop stages and tillage operations. Overland sheet and rill erosion of sediment for each cell is determined using RUSLE (Revised Universal Soil Loss Equation) (USDA, 1996). The sediment transport and deposition are determined using HUSLE (Hydro-geomorphic Universal Soil Loss Equation) and the modified Einstein's equation, respectively (Bingner et al., 2009).

In AnnAGNPS peak flow calculations are performed using TR-55 graphical peak discharge method (NRCS, 1986). A daily mass balance for nitrogen, phosphorous, and organic carbon are calculated for each cell. Both nutrients and pesticides are subdivided into soluble and sediment attached components for routing. Each nutrient component is decayed based upon the reach travel time, water temperature, and an appropriate decay constant (Bingner et al., 2009).

The model was designed to simulate long term sediment and chemical transport within watersheds. A source accounting function is one of the distinctive features of the model. The model may be used to estimate the water, sediment, and chemical loadings at any point as well as the areas contributing at any point in the watershed. One of the outputs from the model is the contribution of each location as a ratio to the loadings at watershed outlet. This feature is of paramount importance in identifying critical areas that are contributing to the flow, sediment and chemicals at the outlet of the watershed. Thus, the model can be used to examine current conditions in a watershed, to compare effects of different conservation alternatives, to evaluate the BMPs and to analyze risks and cost/benefits within a watershed (Yuan et al., 2003 and Bingner et al., 2009).

The basic components modelled in AnnAGNPS are hydrology, sediment, nutrient and pesticide transport. The model requires numerous physical parameters to characterize the watershed incorporating soil data, meteorological records, landuse information and management data. A number of included modules supplied with the AnnAGNPS software package were utilized in the preparation of the AnnAGNPS database.

The watershed variability is approximated in AnnAGNPS using cells that have homogenous properties during the simulation. The cells have the same landuse properties, soil properties, and land management practices within a single drainage area. Figure 3-1 shows the major process emulated in AnnAGNPS watershed simulation. A daily water soil water budget is maintained for each cell; refer to Equation 3-1 as outlined in section 3.3. The cell network representing the properties of a watershed is connected by a network of reaches.

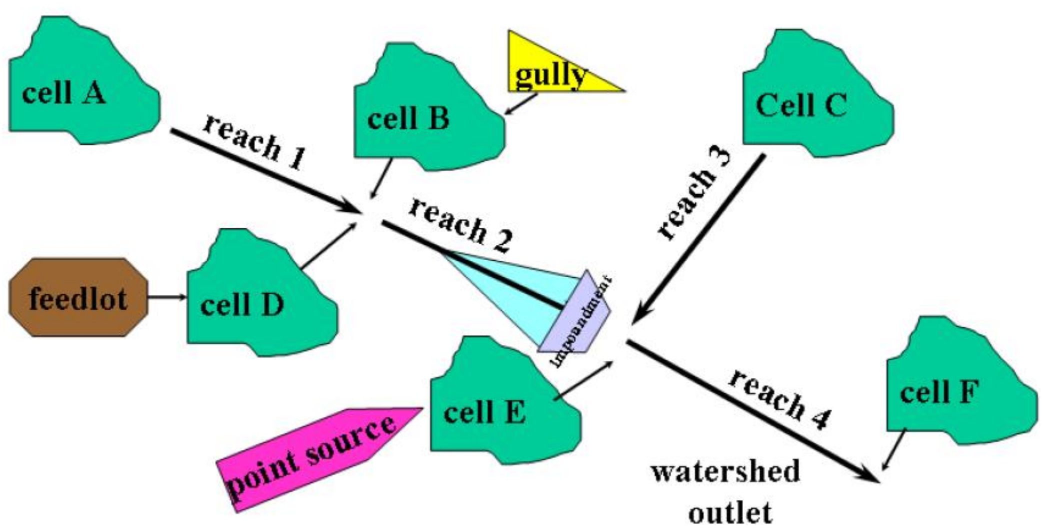

Figure 3-1: AnnAGNPS Major Processes

(Source: Bingner et al., 2009)

The relevant physical parameters of a study watershed, such as cell and stream network information can be extracted from a watershed's digital elevation models (DEM) using the TOPAGNPS module. The AGFLOW module is used to determine the topographic related input parameters for AnnAGNPS and to format the TOPAGNPS output in the form needed by AnnAGNPS (Bingner et al., 2009). These physical parameters of the watershed are held constant throughout the simulation period.

3.3 AnnAGNPS Meteorological Data

To utilize the AnnAGNPS model, a complete chronological set of climate data is required for the entire simulation period. At least one (primary) climate data file is required for the AnnAGNPS model implementation. The primary climate data file must have daily values for all required fields during the simulation period: however, secondary climate files can be included for other cells. If data gaps exist in the secondary climate files the missing data is filled in from the primary climate file dataset.

Six daily weather parameters are needed for the AnnAGNPS simulation; minimum air temperature; maximum air temperature; precipitation; dew point; sky cover; and wind speed. The sky cover data can be replaced with solar radiation at the ground level if available. Weather data is the first fundamental data input necessary to derive output results from the AnnAGNPS model. Daily precipitation is the primary driver of the hydrologic cycle, temperature data is used to define frozen conditions, and the remaining meteorological data is used to determine actual evapotranspiration (AET) (Bingner et al., 2009). The selection of the source of climate data and the years utilized is fundamental for proper simulations with the AnnAGNPS software package.

3.4 AnnAGNPS Water Processes

All non-point source loadings in AnnAGNPS are driven by the hydrologic cycle. In AnnAGNPS water is applied to the land as irrigation or precipitation. There are three irrigation application systems emulated in the software; surface irrigation; sprinkler irrigation; and trickle irrigation. The sediment yield is determined using a separate equation from the one used for precipitation.

The hydrology model is based upon the water balance equation. Equation 3-1, below, is used on a daily time step to determine the soil moisture. The water budget in the AnnAGNPS soil profiles are simulated in two layers. The first soil layer is 203.2 mm in depth, referred to as the tillage layer, as defined by RUSLE. The second soil layer is defined from the bottom of the tillage layer to either an impervious layer or the remainder of the user defined soil profile.

$$SM_{t+1} = SM_t + \frac{WI_t - Q_t - PERC_t - ET_t - Q_{lat} - Q_{title}}{Z} \qquad \text{Equation 3-1}$$

In Equation 3-1 $\mathrm{SM_t}$ represents the moisture content for each soil layer at the beginning of the time period. The $\mathrm{SM_t}$ value is a dimensionless fractional value; characterizing the percent volume of the water in a soil column to the total volume. The $\mathrm{SM_{t+1}}$ represents the soil moisture content at the end of the time period (fractional). The $\mathrm{WI_t}$ represents water input consisting of precipitation, snowmelt or irrigation water (mm). The $\mathrm{Q_t}$ represents the surface runoff from the soil profile (mm). The $\mathrm{PERC_t}$ represents the percolation of water out of each soil layer (mm). The $\mathrm{ET_t}$ component of the equation represents the potential evapotranspiration (mm). The $\mathrm{Q_{lat}}$ represents the subsurface lateral flow (mm). The $\mathrm{Q_{title}}$ represents title drain flow (mm). The final variable in the equation, Z, is the thickness of the soil layer (mm).

3.4.1 AnnAGNPS Direct Runoff

The AnnAGNPS pollutant loading program makes daily time step calculations for most model components. However, because of the strong nonlinear dependence of the rate of percolation and evapotranspiration on soil water content, the soil moisture water budget is calculated using sub-daily time steps (Bingner et al., 2009). The $\mathrm{Q_t}$ in Equation

3-1, surface runoff is calculated using the soil conservation service (SCS) curve number (CN) method (The Natural Resource Conservation Service, was formerly known as the Soil Conservation Service). The user inputs SCS CNs based upon CN identifications which are subdivided between soil group types. The curve number identifications are then assigned to different land use management events and schedules.

The CN inputted by the user are for the SCS CN average value (CN_2). The CN_2 value represents the soil moisture halfway between the dry conditions (CN_1) and wet conditions (CN_3). The SCS CN method accounts for CN variability in soil moisture conditions using the Antecedent Runoff Condition (ARC) (USDA, 1997). Equation 3-2 outlines how surface runoff is calculated in AnnAGNPS using the SCS CN method. In AnnAGNPS, the wilting point and the saturation point are assumed to be effectively ARC I and III, respectively. The wilting point and saturation point are defined as the minimum value of soil storage, and maximum soil capacity in the field (Bingner et al., 2009). In Equation 3-2 Q represents runoff in mm, WI represents water input to soil in mm, and S is a variable of CN associated to water retention, with units of mm. Depth of runoff is determined using the below formula as long as WI is greater than $0.2S$, otherwise the runoff is equal to zero.

$$Q = \frac{(WI - 0.2S)^2}{WI + 0.8S} \qquad \text{Equation 3-2}$$

The value of S is a function of CN as outlined in Equation 3-3. However, the value of S used to determine runoff on any specific day is determined using a more complex function involving the fraction of soil saturation, and weighting factors related to the three ARC values of S. The formula used to determine S_t, the water retention daily parameter is outlined in Equation 3-4.

$$S = 254\left(\frac{100}{CN} - 1\right) \qquad \text{Equation 3-3}$$

$$S_t = S_1\left(1 - \frac{FS_t}{FS_t + \exp(W_1 - W_1 W_2)}\right) \qquad \text{Equation 3-4}$$

In Equation 3-4 the S_1 represents the water retention variable associated with ARC I. Both W_1 and W_2 are weighting factors which are a function of S for all three ARCs; the factors formulas are explicitly summarized in the AnnAGNPS Technical Processes (Bingner et al., 2009). The two weights are constant properties for a single CN. However, the FS_t value, the fraction of saturation of the two layer soil system varies daily based upon soil moisture, wilting point, and soil layer depth. The formula for FS_t is also outlined in the AnnAGNPS Technical Processes (Bingner et al., 2009).

3.4.2 AnnAGNPS Evapotranspiration

Potential evapotranspiration a fundamental component of the hydrologic budget is calculated in AnnAGNPS using the Penman Equation, as delineated in Equation 3-5. The potential evapotranspiration is the maximum potential evaporation and transpiration.

$$ET_P = \frac{1}{H_v\left\{\left(\frac{\Delta}{(\Delta+\gamma)}\right)(R-G) + \left(\frac{\gamma}{\Delta+\gamma}\right)W(e_{sat}-e)\right\}} \qquad \text{Equation 3-5}$$

The ET_P in Equation 3-5 represents the potential evapotranspiration in mm. H_v is the latent heat of vaporization in MJ/kg. The Δ, in the same equation, represents the slope of the saturation vapour pressure-temperature curve in kPa/°C. The Δ is a function of temperature and the saturated vapour pressure. R and G are the net radiation and the soil heat flux, respectively. Where preceding both variables have units of MJ/m2. e_{sat} and e represent the saturated vapour pressure and the actual vapour pressure, in kPa. The W represents the wind function as defined by the original Penman wind function, valid from a height of two meters above ground (Jensen et al., 1990).

3.3.3 AnnAGNPS Subsurface Flow Processes

AnnAGNPS only estimates certain components of subsurface flow being lateral subsurface flow or tile drain flow. The amount of lateral flow and tile flow taken from each cell is added to the reach at the same time as runoff and both are considered as the quick return flow (Bingner et al., 2009). The subsurface flow percolation processes modelled in AnnAGNPS are of particular importance in flat regions (Yuan et al., 2006). Lateral subsurface flow is calculated using the one-dimensional Darcy's equation, as outlined in Equation 3-6. Only the saturated case is considered when estimating lateral subsurface flow.

$$q_{lat} = -\frac{K_s\, dh}{dl} \qquad \text{Equation 3-6}$$

Subsurface flow is a very complex process; however Darcy's equation is a commonly used and accurate estimate (Bingner et al., 2009). The relatively simple equation approximates, q_{lat}, the subsurface lateral flow in mm/time period (where the time period is defined by the modeller). The saturated hydraulic conductivity, K_s, is defined for each layer of soil in a cell in mm/time period. The hydraulic gradient, $\frac{dh}{dl}$, is a dimensionless quantity and represents the change in hydraulic head over the change in length. To determine the total volumetric outflow from subsurface lateral flow, q_{lat}, is simply multiplied by lateral flow across area.

Tile drain flow in AnnAGNPS is estimated using the Houghoudt equation, Equation 3-7. To simplify calculations it is assumed that tile flow is steady state; a constant flow and head, where discharge is equivalent to recharge. The Houghoudt equation was selected to simulate surface drainage in AnnAGNPS because of its wide applicability and relatively simple structure (Yuan et al., 2006). Figure 3-2, outlines a general form of the Houghoudt tile flow. The direction of flow is perpendicular to the diagram, where horizontal flow travels towards the tile drains. The water table above parallel drains is normally approximated using an elliptical shape (Bingner et al., 2009).

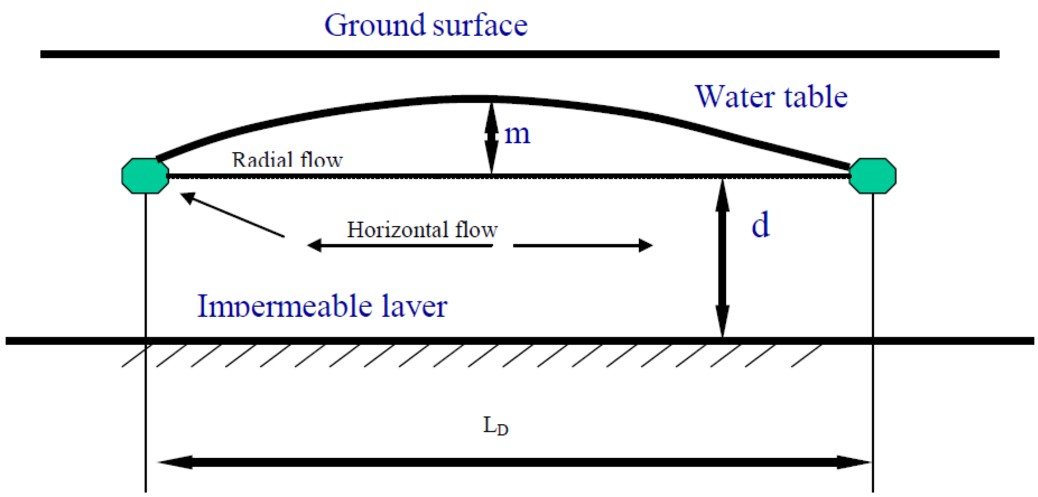

Figure 3-2: Schematic for Houghoudt Tile Flow

(Source: Bingner et al., 2009)

$$q_{drain} = \frac{8K_s d_e m + 4K_s m^2}{L_D^2} \qquad \text{Equation 3-7}$$

The drainage flux, q_{drain}, is defined as the tile flow from a cell in units of mm/time period. The saturated hydraulic conductivity is the same as defined for Equation 3-6. The equivalent depth of the impermeable layer below the drain, d_e, is defined by a series of equations that are a function of the flow geometry, as outlined in Bingner et al. (2009). The midpoint water table height above the drain, m, is outlined in Figure 4-2. The distance between the drains, L_D, is also outlined in Figure 4-2. The units for d_e, m, and L_D are meters. The depth of the soil saturation from the impervious layer is fundamental in determining the tile drain flow (Yuan et al., 2006). The total volumetric tile drain flow from a cell is simply q_{drain} multiplied by area of the cell.

3.4.4 AnnAGNPS Channel Hydraulics and Hydrology

AnnAGNPS watershed simulations are composed of two primary components; cells and reaches. Both components have different methodologies for determining travel time and peak flow rates. The in cell water transport processes can be subdivided into three sections; overland flow, shallow concentrated flow, and concentrated flow. Whereas the reach water transport process is emulated by a single process concentrated flow.

The concentrated in-cell flow is assumed to have a trapezoidal cross-section. The depth of flow is solved for using Newton's method and Manning's equation with the assumption that the wetted parameter is equal to the top width. The procedure for the solution is outlined in Bingner et al. (2009). The time of concentration for cells and the travel time from a channel reach are calculated in AnnAGNPS, and are needed to calculate peak water discharge and pre-peak runoff fraction with the extended TR-55 methodology (NRCS, 1986).

The time of concentration for in cell flow is approximated in AnnAGNPS as the sum of the travel times for in-cell overland flow, shallow concentrated flow, and concentrated flow. The total length of the in-cell flow path is defined as an input for the Arcview-AnnAGNPS cell delineation. This total travel length is the sum of the three lengths for each flow process. In AnnAGNPS overland flow is the first segment of flow, shallow concentrated flow is the second segment of flow and concentrated flow is the last segment. The methodology for determining all three travel times are outlined in Bingner et al. (2009).

To determine travel time in an individual reach, Equation 3-8 is utilized. The methodology is much simpler that for cells. The travel time through a reach segment, $T_{t,reach}$, is in hours. The velocity of the flow through the reach, V_{Reach}, is in m/s. The ΔL in Equation 3-8 simply represents the length of the reach segment.

$$T_{t,reach} = \frac{\Delta L}{3600 \times V_{Reach}} \qquad \text{Equation 3-8}$$

The time of concentration at any reach outlet is defined as the maximum sum of the times of concentration from all contributing reaches plus the travel time through the current reach. The peak discharge in an individual cell is a function of the rainfall distribution type as defined in TR-55 (NRCS, 1986), initial abstraction (I_a), and the effective depth of the 24-hour precipitation (P_{24}). The initial abstraction is defined as 20 percent of water retention parameter, S, as previously noted in Equation 3-3.

Equation 3-9 outlines the formula used to determine the peak discharge rate, Q_p, for a cell in AnnAGNPS. The formula was determined using a set of regression coefficient using the TR-55 extended procedure and curve fitting tools. In Equation 3-9, the variable D_a is defined as the total drainage area generating the peak flow rate with units of hectares. The variable T_c is defined as the time of concentration for in-cell flow with units of hours. The variables $a, b, c, d, e,$ and f are all regression coefficients which are a function of I_a/P_{24} and rainfall distribution type. Tables for the regression coefficients are outlined in AnnAGNPS Technical Processes (Bingner et al., 2009).

$$Q_p = 2.7777778 \times 10^{-3} \times P_{24} \times D_a \times \frac{a+(c\times T_c)+\left(e\times T_c^2\right)}{1+(b\times T_c)+\left(d\times T_c^2\right)+(f\times T_c^3)} \qquad \text{Equation 3-9}$$

The hydrograph of flow from the cells in AnnAGNPS is triangular. This simple hydrograph is assumed to be sufficient for modelling proposes. Since the sediment transport processes are only concerned with the duration for an average discharge, the time to peak is not important and a right angled triangular hydrograph is used to calculate sediment transport (Bingner et al., 2009).

3.5 AnnAGNPS Sediment Processes

The sediment erosion and transport processes driven by the hydrologic cycle in a watershed are complex natural systems emulated by AnnAGNPS. AnnAGNPS is capable of modelling four types of erosion; rill erosion, sheet erosion, gully erosion, and stream bed erosion (see Figure 3-3). AnnAGNPS simulates both sediment transportation across cells and in streams (reaches).

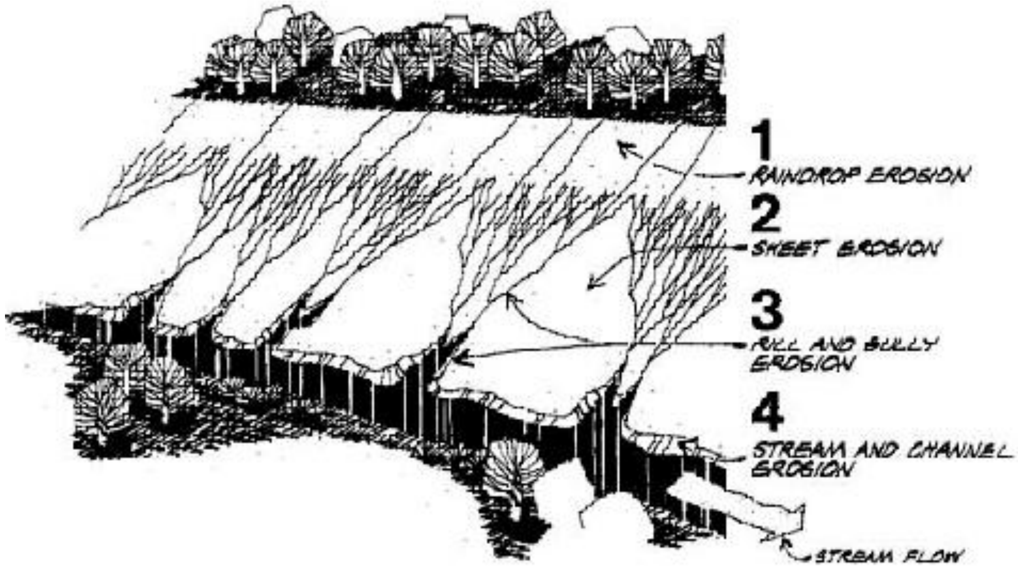

Figure 3-3: Erosion Processes

(Source: USDA, 2009)

In AnnAGNPS RUSLE, the Revised Universal Soil Loss Equation, is used to determine sheet and rill sediment erosion. RUSLE is based upon the USLE, the Universal Soil Loss Equation, both of which were developed by the USDA to determine land erosion caused by interaction of rainfall/runoff with the soil. The two equations have several similarities including the form of the equations (see Equation 3-10). The USLE was published in 1965 in the USDA Agricultural Handbook 282 and 31 years later, in 1996 the RULSE was published in the USDA Agricultural Handbook 703.

$$A = R \times K \times LS \times C \times P \qquad \text{Equation 3-10}$$

In AnnAGNPS, version 1.05 of the RUSLE software is utilized to estimate sheet and rill erosion. Equation 3-10 is used to calculate erosion of a certain soil and landuse where A represents the average annual soil loss, R is a rainfall and runoff factor, K the soil erodibility factor, L is a slope length factor, S is a slope steepness factor, C is a cover and management factor, and P is a support practice cover factor. The units of A are often ton/acre/yr but other units can be used if the corresponding factor values are altered.

The R factor is proportional to the intensity and duration of a storm-event where higher values correspond to higher erosion potential. The values for R are often characterized for distinct geographically regions. The K factor is the average soil loss in for a particular soil in cultivated, continuous and slope steepness of 9%. Additionally, this factor represents a measure of the susceptibility of soil particles to detachment and transport by rainfall (Stone and Hilborn, 2011). Several soil properties dictate the value of K including texture, structure, organic content and hydraulic conductivity.

It is common to group the effects of land slope, S, and land length, L, into a single factor LS. This combination factor is commonly referred to as the slope length gradient factor. As the LS factor increases so does the potential erosion from either an increased slope or an increase in the length of a sloped section.

The C and P factors are both related to the landuse management practices. The C factor is commonly known as the crop, vegetation and management factor. The C factor is a function of the relative effective of soil and crop management systems in the prevention of erosion. The C factor is also a ratio of the soil loss from land under a specific crop and management system to the corresponding loss from continuously fallow and tilled land. The P factor's primary purpose is to emulate the effects of runoff control practices; any practice that reduces the volume or rate of runoff is controlled by a corresponding P. The P factor represents the ratio of soil loss by a support practice to that of straight-row farming.

The C, P, and K factors are all calculated for each cell on a 15 day period before the simulation is run. These factors are based on landuse practices implemented and soil properties. The LS, slope length gradient, factor is inputted by the user for each cell. The R, rainfall and runoff, factor is calculated for each rainfall event, and is based on a user defined maximum energy.

The sediment erosion and transport particles are segregated into 5 classes (shown in the Table 3-1). Each soil type and layer has its own unique percent composition of clay, silt, sand, small aggregates, and large aggregates.

Table 3-1: AnnAGNPS Sediment Particle Properties

Particle-size Name	**Size (mm)**	**Particle Density (Mg/m^3)**	**Fall Velocity (mm/s)**	**Equivalent Sand Size (mm)**	**Deposition Rate Ratio**
Clay	<0.002	2.6	3.11E-03	2.00E-03	0.000091
Silt	0.002-0.050	2.65	8.02E-02	1.00E-02	0.002401
Sand	0.050-2.000	2.65	2.31E+01	2.00E-01	0.691528
Small Aggregates	0.020-0.075	1.8	3.81E-01	3.51E-02	0.007747
Large Aggregates	0.200-1.000	1.6	1.65E+01	5.00E-01	0.298233

(source: Bingner et al., 2009)

Gully erosion yield in AnnAGNPS is calculated using the Revised Empirical Gully Erosion Model (REGEM) which was developed by the USDA to estimate gully erosion (Bingner et al., 2009). There are three major causes of gully erosion (1) an increase in surface water flow, (2) a decrease in soil erosion resistance, and (3) a constant saturation of the soil (Gordon, et. al, 2007). The science behind understanding and quantifying gully erosion is still under development with considerable work to be done. Currently, the REGEM system in AnnAGNPS uses the largest shear stress of five different possible equations to predict how much gully erosion will occur and the new size of the gully opening in the future time step.

The sediment that is transported overland is determined by HUSLE, the Hydro-geomorphic Universal Soil Loss Equation. HUSLE calculates the delivery ratio of sediment yield based upon the erosion calculated by RUSLE. Equation 3-11, taken from Bingner et al. (2009), is the sediment yield (S_y) in Mg/ha, from a given source area. In Equation 3-11, Q represents the total runoff depth in mm, q_p represents the peak rate of surface runoff, and the K, L, S, C, and P factors are the same as previously defined for

RUSLE. The interpretation of the delivery ratio is the sediment yield at one location divided by any other location.

$$S_y = 0.22 * Q^{0.68} * q_p^{0.95} * K * LS * C * P \qquad \text{Equation 3-11}$$

The final component to sediment loading in AnnAGNPS is the stream erosion and sediment transport. The stream erosion and transport are dependent so they are discussed as one topic. The stream bed erosion system in AnnAGNPS is simply a function of stream sediment loading capacity for each size class. If the channel has a particle type and current loading is below stream capacity and has the appropriate shear stress, then sediment load downstream will be equal to capacity. However, if the channel has a particle type and current loading is above stream capacity, then sediment load deposition occurs downstream.

To determine the quantity of downstream sediment loading a modified Einstein's equation is used. Equation 3-12, taken from Bingner et al. (2009) explains this system. The downstream and upstream sediment loads are q_{s2} and q_{s1}, respectively. Additionally, the deposition number, Einstein's proportionality constant and sediment capacity can all be solved implicitly in each time step. In Equation 3-12 N_d is the deposition number and is a function of Einstein proportionality constant (A_e). Finally q_{sc} is the sediment capacity for particle size c.

$$q_{s2} = q_{sc} + (q_{s1} - q_{sc}) * exp(-N_d) \qquad \text{Equation 3-12}$$

3.6 AnnAGNPS Nutrient and Pesticide Processes

The AnnAGNPS model is capable of simulating an unlimited number of pesticides and three nutrients processes. The pesticides in AnnAGNPS are assumed to have independent chemical properties; however, each pesticide is treated separately where independent equilibrium for each is assumed (Bingner et. al, 2009). The three chemicals that are emulated in AnnAGNPS are nitrogen, phosphorous, and organic carbon. Both nitrogen and phosphorous are modelled in their soluble and absorbed states. Organic carbon also only exists in an insoluble phase, attached to clay sized particles. Mass balance calculations are preformed for both absorbed and dissolved chemicals occurring at the end of each stream reach. A re-equilibrium process for both absorbed and dissolved chemicals is calculated at the downstream end of each reach if clay sized particles are deposited or if there is a loss of water.

3.6.1 AnnAGNPS Nitrogen Processes

The mass balance for nitrogen is the most complex of the three nutrients modelled in AnnAGNPS. The nitrogen processes of losses and gains are numerous; including mineralization, fixation, fertilization, plant uptake, denitrification, volatilization, and immobilization (Bingner et al., 2009). The nitrogen processes modelled in AnnAGNPS are a simplification of the actual processes occurring in nature. Major components considered are the uptake of nitrogen by plants, the application of fertilizers, residue decomposition, and runoff movement of nitrogen (Yuan et al., 2003).

The user defines the initial amount of inorganic and organic nitrogen in the simulated soils. Equation 3-13 is used as a conversion factor to convert nutrient concentration in a soil to kilograms. In the same equation $conv$ represents an intensive unit to extensive unit conversion factor, with units of kilograms. The D represents the thickness of a soil layer, with units of mm. The ρ_b represents bulk density of the composite soil layer with units of tons/m^3. Lastly, the A_{cell} variable represents the cells area with units of hectares.

$$conv = D * 10 * 1000 * \rho_b * A_{cell} \qquad \text{Equation 3-13}$$

In AnnAGNPS the organic nitrogen mass balance is maintained on an individual cell basis for both soil layers (the first 203.2 mm, the tillage layer, and the remainder). The mass balance equations for all nutrients, organic and inorganic, have a similar form.

In Equation 3-14 the organic nitrogen mass balance formula is outlined. In the same equation $orgN_t$ and $orgN_{t-1}$ represent the concentration of total organic nitrogen in ppm for the current and previous day, respectively. The $resN$ represents organic nitrogen additions from the decomposition of crop and non-crop residue laying on the soil surface to the cell upper layer on the current day, with units of kilograms. The fer_orgN represents the organic nitrogen input from fertilizer applications, with units of kilograms. The $hmnN$ variable represents the loss of organic nitrogen from the soil that is mineralized to inorganic nitrogen on the current day, with units of kilograms. The $sedN$ variable represents the current day's mass of organic nitrogen attached to sediment, in units of kilograms. The calculation of all variables mentioned in Equation 3-14 are explained in greater detail in AnnAGNPS TECHNICAL PROCESSES: Documentation (Bingner et al., 2009).

$$orgN_t = orgN_{t-1} + \frac{(resN + fer\_orgN - hmnN - sedN) * 1000000}{conv} \qquad \text{Equation 3-14}$$

The inorganic nitrogen mass balance equation follows a similar format as the organic nitrogen mass balance. In Equation 3-15 the inorganic nitrogen mass balance formula is outlined. In the same equation $inorgN_{i+1}$ and $inorgN_i$ represent the concentration of total inorganic nitrogen in ppm for the current and previous day, respectively. The $hmnN$ variable represents the daily inorganic input of mineralized nitrogen from organic matter, with units of kilograms. The $uptN$ variable represents the amount of inorganic nitrogen uptake of a plant in a growth stage, with units of kilograms. The $cell\_soil\_sol\_N$ variable represents the soils lost of inorganic nitrogen to runoff, with units of kilograms. This inorganic nitrogen lost to runoff, includes modelled direct surface runoff, leaching, and lateral subsurface flow. The DN variable represents the denitrification output of inorganic nitrogen from the soil, with units of kilograms. The $conv$ variable is the same as defined in Equation 3-13.

$$inorgN_{i+1} = inorgN_i + \frac{(hmnN - uptN - cell\_soil\_sol\_N - DN) * 1000000}{conv} \qquad \text{Equation 3-15}$$

The nitrogen model in AnnAGNPS is simplification of the real world and an empirical methodology is used for the simulation (Bingner, et al, 2009). Nitrogen fixation and volatilization are not components of the AnnAGNPS model (Yuan et al., 2003). Nitrogen fixation and volatilization both involve the transformation of nitrogen from a gas to another form, or from another form to a gas, respectively. These atmosphere processes are not modelled in AnnAGNPS. However, these components of the nitrogen budget may be significant. In the United States, nitrogen fixation produces about one third of the amount of fertilizer applied (Havlin et al., 1999).

3.6.2 AnnAGNPS Phosphorous Processes

Similar to the nitrogen modelling components in AnnAGNPS, a separate mass balance for both organic and inorganic phosphorous is maintained (Yuan, et al., 2005). The AnnAGNPS model for phosphorous processes is a simplification of the natural systems where some components are ignored and included components are determined using an empirical approach (Bingner, et al., 2009).

The phosphorous mass balance in AnnAGNPS cells occurs in two distinct layers: the upper tillage layer (up to a depth of 203.2 mm), and the remainder of the user defined soil layer. The equation used to simulate the organic phosphorus mass balance in the tillage layer is outlined in Equation 3-16. There are three major pools of phosphorous accounted for in AnnAGNPS; active, stable and fresh (Yuan, et al., 2005).

$$orgP_t = orgP_{t-1} + \frac{(resP + fer\_orgP - hmnP - sed\_orgP) * 1000000}{conv} \qquad \text{Equation 3-16}$$

In the same equation $orgP_t$ and $orgP_{t-1}$ represent the concentration of total organic phosphorous in ppm for the current and previous day, respectively. The $resP$ represents organic phosphorous additions from the decomposition of rest crop residue on the current day, with units of kilograms. The fer_orgP variable represents the organic phosphorous input from fertilizer applications, with units of kilograms. The $hmnP$

variable represents the loss of organic phosphorous from the soil that mineralized to inorganic phosphorous on the current day, with units of kilograms. The sed_orgP variable represents the current day's mass of organic phosphorous attached to sediment, in units of kilograms. The calculation of all variables mentioned in Equation 3-16 are explained in greater detail in *AnnAGNPS TECHNICAL PROCESSES: Documentation* (Bingner et al., 2009).

Similar to organic phosphorous AnnAGNPS also monitors three different pools of inorganic phosphorous; active, stable and solution (Yuan et al, 2005). The total soluble (inorganic) phosphorous outflow from a cell is outlined in Equation 3-17. This loss, sol_P, represents the total mass (kg) of inorganic phosphorous lost to from the soil surface and loss from the composite soil layer. The variables $cell\_soil\_sol\_P$ and surf_sol_P represent the nitrogen losses to runoff from the composite soil layer and nitrogen losses to runoff from the soil surface, respectively.

$$sol\_P = cell\_soil\_sol\_P + surf\_sol\_P \qquad \text{Equation 3-17}$$

Unlike the nitrogen modelling, AnnAGNPS does not simulate the leaching or subsurface lateral flow movement of soluble phosphorous (Bingner et al., 2009). This is due to the low mobility of phosphorous. In AnnAGNPS only surface runoff transports phosphorous from a cell.

3.6.3 AnnAGNPS Organic Carbon and Pesticide Processes

The discussion on pesticide transport is limited in the AnnAGNPS TECHNICAL PROCESSES: Documentation (Bingner et al., 2009). The pesticide model simply follows a mass balance procedure: there is no interaction between individual pesticides, and the pesticides' properties are inputted by the user.

The organic carbon processes emulated by AnnAGNPS follow a similar structure as those of other nutrients simulated by the model. The organic carbon in-cells is accounted for using a mass balance procedure. The organic carbon in a cell is accounted for using Equation 3-18. In the same equation $\mathrm{orgC_t}$ and $\mathrm{orgC_{t-1}}$ represent the total organic carbon, as a faction, for the current and previous day, respectively. The $resC$ represents organic carbon additions from the decomposition of crop and non-crop residue laying on the soil surface to the cell upper layer on the current day, with units of kilograms. The fer_orgC represents the organic carbon input from fertilizer applications, with units of kilograms. The $hmnC$ variable represents the loss of organic carbon from the soil that mineralized to inorganic carbon on the current day, with units of kilograms. The $sedC$ variable represents the current day's mass of organic carbon attached to sediment, in units of kilograms. The calculation of all variables mentioned in Equation 3-18 are explained in greater detail in AnnAGNPS TECHNICAL PROCESSES: Documentation (Bingner et al., 2009). Only organic carbon is calculated in AnnAGNPS. A mass balance for inorganic carbon is not maintained.

$$orgC_t = orgC_{t-1} + \frac{(resC + fer\_orgC - hmnC - sedC)}{conv} \qquad \text{Equation 3-18}$$

3.7 Summary

This chapter provides a summary of the major systems emulated in AnnAGNPS. For a more detailed review of the model's systems review AnnAGNPS Technical Processes (Bingner et al., 2009).

CHAPTER 4: BIG CREEK MARSH WATER BUDGET MODEL DESCRIPTION

4.1 Introduction

The following chapter provides an outline of the Big Creek Marsh wetland's theoretical historical water budget (WB) model. The Big Creek Marsh WB model approximates the hydrological processes within the wetland ecosystem including both natural and anthropogenic flows. In this chapter the development of the WB model will be highlighted step by step. The chapter will discuss the major logic assumptions both in the general model structure and in individual model components. The data utilized in the model construction will also be outlined in the chapter.

A WB is a mathematical model of a mass balance of water entering and leaving a system. A review of WBs, with an emphasis on wetlands, is contained in Chapter 2. When performing a WB several components must be selected including a control volume, inflows and outflows which are deemed significant, and appropriate physical and empirical models that emulate their respective components (ERCA, 2011a). Understanding the hydrological components is fundamental to the management, restoration, and monitoring of a wetland ecosystem as it provides insight into the health, baseline conditions and variability of the same ecosystem (Erwin, 2009).

WB models can be developed for any geographical region with significant variance in spatial and temporal scales. Wetland WBs provide unique challenges. These challenges often stem from the great variable between individual wetlands and even the variability within a single wetland. The great variability between individual wetlands includes different meteorological patterns, land characteristics, soil properties, vegetation, and streamflow regimes. This variability of wetlands necessitates a broad definition requiring that all or some of the following be found in a wetland: periodic or seasonal flooding, the presence of a surface or near surface ground water table, hydric undrained soil, and the growth of hydrophytes (wetland vegetation) are promoted (Tiner, 1999).

4.2 History and Background of Big Creek Marsh

Big Creek Marsh is a riparian wetland, in Essex County, Ontario, Canada (Figure 4-1). The wetland is fed by streamflow from the greater Big Creek Watershed. The Marsh is approximately 682 hectares in area and is located east of the Detroit River, and north of Lake Erie to which it also outlets (Ducks Unlimited Canada, 2007). The water levels in the Marsh have been artificially managed by land owners since 1909 using a system of pumps and a hydraulic control structure. Near the outlet of the Marsh the wetland has wide shallow open waters resembling a fresh water estuary and is separated from Lake Erie by a weir, narrow barrier beach and a low, fore dune complex stabilized by vegetation (Wilson and Cheskey, 2000).

The wetland exhibits primarily marsh wetland characteristics with some, approximately ten percent, swamp wetland characteristics (Waldron, 1998). Typical features of marsh wetlands include periodic or permanent flooding, the growth of non-woody plants (shrubs, reeds, cattails, and water lilies), and open expanses of slow moving or standing water (LandOwner Resource Centre, 1997). Typical features of swamp wetlands include seasonal or permanent flooding, the growth of woody plants (trees and shrubs), and are often dry in mid to late summer months (Environment Canada, 1997).

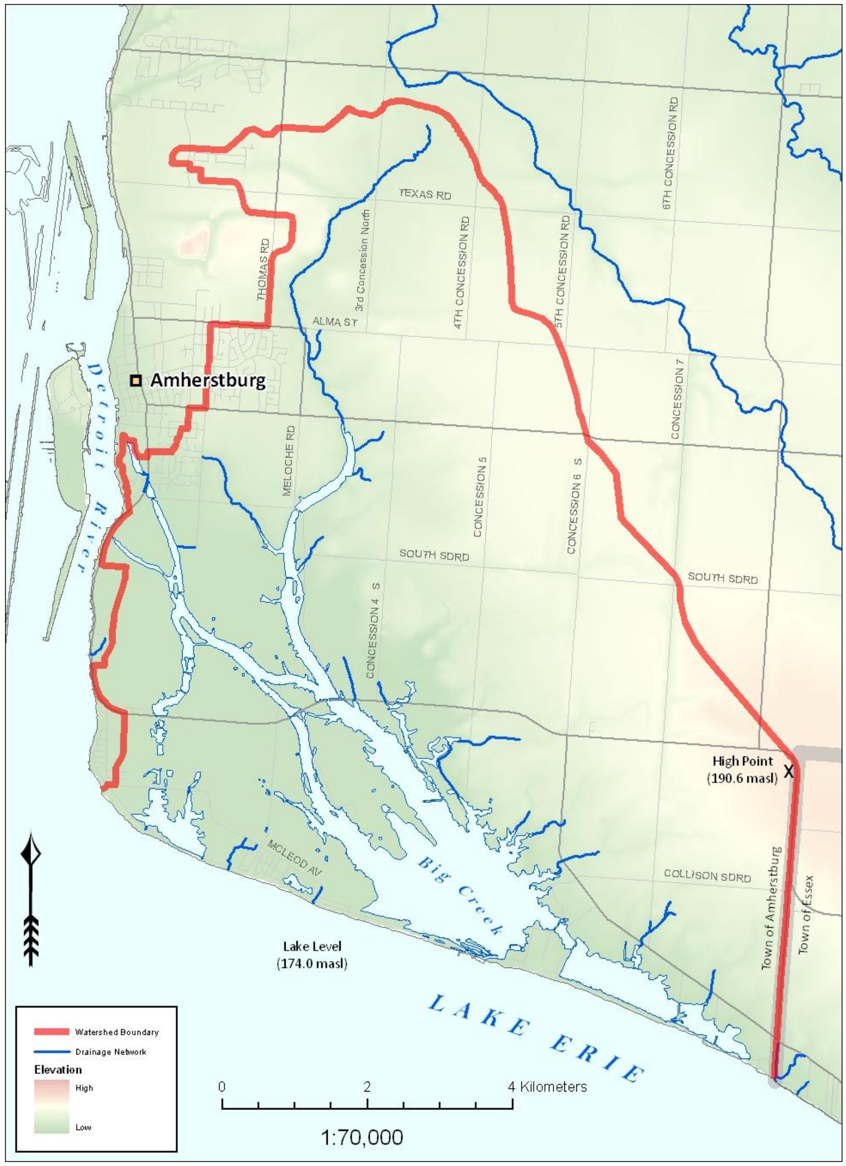

Figure 4-1: Big Creek Marsh

(Source: ERCA, 2011a)

Big Creek Marsh is considered an important waterfowl staging area. Select rare plant species residing in Big Creek include the American Lotus, Prairie White Fringed-orchid and Swamp Rose Mallow (Waldron, 1998). Additionally, rare animal species that can be found in Big Creek include the Eastern Fox Snake, the Spotted Turtle, the Prothonotary Warbler and the Bald Eagle (Wilson and Cheskey, 2000). Big Creek Marsh is identified as an Area of Natural and Scientific Interest (ANSI), an

Environmentally Significant Area (ESA), and a globally Important Bird Area (IBA) (ERCA, 2008).

Big Creek Marsh, like other wetlands, provides countless and often intangible values. The values or functions that wetlands as a resource provide are often difficult to quantify and consequently is the assessment of their monetary value. These functions include wildlife habitat, water pollution control, sediment control, groundwater recharge, flood storage, erosion control, educational value, recreational value, and aesthetic value (LandOwner Resource Centre, 1997). Essex County, before European development in the 1800's, was primary composed of swampy watersheds (Waldron, 1998) with slower flow rates and larger storage capacities. For urbanization and settlement purposes artificial drainage systems were implemented that drastically modified the ecological and hydrological characteristics of the region. The Big Creek Marsh should not only be preserved for its functions but also for its historic ecological significance.

4.3 Structure of the Big Creek Marsh Water Budget

The following section outlines the general structure of the water balance. Before any components of the budget were determined, an outline of the mathematical model used to estimate the physical processes occurring in the Marsh was first conceptualized. Consequently, this section will outline the general model structure first; then discuss the equations and assumptions used to determine each individual WB component.

The hydrological processes that occur in wetlands are essentially the same that occur outside of wetlands. The major components of the hydrologic cycle that should be considered include surface water flow, precipitation, groundwater flow, and evapotranspiration (USGS, 1999). In contrast to other regions both a favourable geologic setting and an adequate supply of water are necessary for a wetland's existence. The Wetland Reserve Program (WRP) supported by the United States Department of Agriculture (USDA) Natural Resource Conservation Service has produced a small series of literature that agree with the USGS paper (1999) regarding the components to include in a wetland WB.

The WRP technical notes (1993a and 1993b) describe the fundamental components affecting a wetlands hydrologic balance. These components are summarized in the following paragraph. A defining feature of wetland WBs are the basin characteristics; considering both the wetland's geographic features and the features of the watershed. Precipitation is crucial in the budget creating an inflow directly through rain occurring within the wetland's physical limits and as the source of streamflow generation. In wetland WBs the affect of evaporation and transpiration should be integrated. The culminations of the two outflows are often referred to as evapotranspiration (ET). The magnitude of groundwater recharge and discharge can be significant components of the balance. Additionally, tides or other coastal processes are outlined as components that can radically alter wetland storage.

A WB is a mass balance on the net flux of water entering and leaving a system or control volume. For the Big Creek Marsh WB inflow components were assumed to be effective precipitation (R), streamflow from the Big Creek Watershed (SF), seepage from Lake Erie into Big Creek (Sin), water pumped in from Lake Erie to maintain the desired water level (P_{in}), and flow overtopping the control dam structure entering the Marsh (OF_{in}). Groundwater inflows from sources excluding Lake Erie are neglected. Outflow components that were considered include evapotranspiration (ET), seepage flow to Lake Erie (S_{out}), pumping out of the Marsh (P_{out}), outflow to Lake Erie from the control gate (G), and flow overtopping the control dam structure from the Marsh (OF_{out}). Similarly groundwater outflow to sources excluding Lake Erie are neglected. For clarification purposes in this report seepage refers to the exchange of water between the Lake and Marsh.

The disregard of deep groundwater flux may cause an error in the water budget estimate; however, for the purpose of this report it is assumed that inter-watershed groundwater inflow is roughly equivalent to groundwater outflow. The daily change in storage (ΔS) was calculated using Equation 4-1. The new total storage was determined by adding the change to the previous time step's total storage.

$$\Delta \mathrm{S} = \mathrm{R} + \mathrm{SF} + \mathrm{S_{in}} + \mathrm{P_{in}} + \mathrm{OF_{in}} - \mathrm{G} - \mathrm{S_{out}} - \mathrm{ET} - \mathrm{OF_{out}} - \mathrm{P_{out}} \quad \text{Equation 4-1}$$

4.3.1 Wetland Operating Practices

Limited data of the operating practices in Big Creek Marsh exist, however; three operating scenarios are outlined in the permit to take water (PTTW) for the wetland (Ducks Unlimited Canada, 2007). The outlined operating scenarios are the hemi phase, open water phase, and the overgrown phases. The target marsh depths for each month and phase are outlined in Table 4.1. The depths in the same table are measured from a datum of 173.7 metres above mean sea level (AMSL). To maintain the different water levels in each of the three operating phases distinct patterns of pumping into or out of the Marsh vary seasonally. The maximum total daily and annual flow pumped into the Marsh is outlined in the PTTW (Ducks Unlimited Canada, 2007).

Table 4-1: Target Marsh Depths (173.7 m AMSL)

Month	**Hemi Marsh Phase**	**Open Water Marsh Phase**	**Overgrown Marsh Phase**
January	0.4	0.28	0.6
February	0.4	0.27	0.6
March	0.4	0.35	0.6
April	0.5	0.3	0.7
May	0.6	0.2	0.8
June	0.7	0.15	0.9
July	0.7	0.15	0.9
August	0.7	0.35	0.9
September	0.8	0.55	0.8
October	0.8	0.65	0.8
November	0.8	0.65	0.8
December	0.8	0.65	0.8

The three phases represent different wetland water level annual scenarios. The overgrown phase corresponds to generally higher water levels in the Marsh (a wet year). The hemi phase corresponds to intermediate water levels (an average year). The open water phase corresponds to generally lower water levels in the Marsh (a dry year). The phases are closely linked to the ecosystem health and fluctuations within a wetland. The overgrown phase with high water levels floods out existing vegetation to provide space for new growth (Ducks Unlimited Canada, 2007). The open water phase with low waters allows for germination of new plants in a wetland (Paveglio and Kessler, 2004). The hemi phase is generally considered the most ecologically productive phase of the three supporting the maximum bio-diversity in a wetland normally with a 1:1 ratio of open water and emergent vegetation (U.S. Army Corps of Engineers, 2009).

The three phases work as a natural re-growth cycle where the overgrown phase exterminates existing vegetation, the open water phase promotes the growth of new plants, and the hemi phase enables the support of the most number of species within the ecosystem. Historic records outlining which phase was implemented in each calendar year were not available. However, three years (2006-2008) of recorded pumping flow into the Marsh was available and could be used to determine which operating phase was most likely in a given year. The recorded pumping data is reviewed in section 4.3.4.

4.3.2 Big Creek Marsh Bathymetry and Storage

The change in storage in the Big Creek Marsh is determined using Equation 4.1; however, to estimate other components of the WB additional geometry of the Marsh and of the water stored in the Marsh are necessary. Only one flow outlined in Equation 4.1 is determined independently of the Marsh geometry, the streamflow entering the Marsh. All other variables are a function of the Marsh total area, open water area, or water depth. Three bathymetric surveys of the Marsh were utilized to estimate the relationship of the wetland storage to depth. Two surveys existed before the research work and one was conducted in the western leg of the Marsh to obtain additional data.

Utilizing the three data sets the surface area of the Marsh was estimated with different depths of water, using a 0.1 meter step. The relationship of surface area to depth was then used to estimate the storage-depth curve (Equation 4.2) using linear interpolation. The datum for the storage-depth curve is 173.7 meters above sea level. The total live storage in the Big Creek Marsh is approximately 4.5 million m^3 before water will overtop the dam structure.

$$S(h) = 3{,}430{,}030 * h^{1.48142} \ \{\text{were h is in m \& S is in } m^3\} \qquad \text{Equation 4-2}$$

4.3.3 Precipitation, Streamflow, and Gate Controlled Flow

The three primary surface flow processes that are considered in the Big Creek Marsh WB are precipitation, streamflow, and gate controlled outflow from the wetland. Each of the three components is calculated using different methodologies.

Precipitation inflow into the Marsh was one of the simpler WB components to estimate. The depth of precipitation (R) inflow was approximated using a simple average of the daily precipitation data from two stations: the Amherstburg (Climate ID: 6130257) and the Harrow (Climate ID: 6133360, 613CC60, 613ZZZZ, & 6133362) weather stations. The total daily depth of precipitation was multiplied by the control volume surface area to determine a daily volumetric system input. This methodology for approximating precipitation inflow has been successfully implemented in several wetland WB studies (Gehrels and Mulamoottil, 1990; Mitsch and Reeder, 1992; Owen, 1995). The monthly average precipitation data for the two gauging stations over the forty year study period is outlined in Figure 4-2. The total precipitation data for each month is outlined in the appendix.

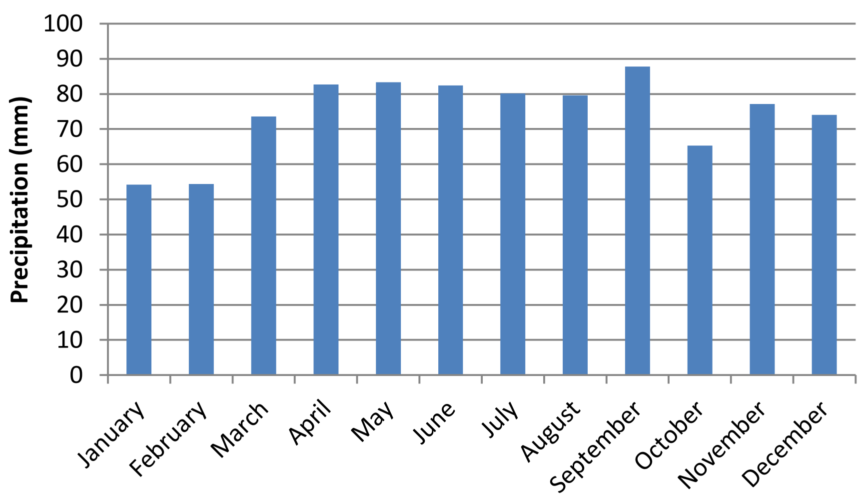

Figure 4-2: Average Monthly Precipitation (mm)

To estimate the magnitude of streamflow (SF) entering the Marsh from all of the wetland's inlets a SWAT model for the Big Creek Watershed was utilized. The SWAT model was delineated into subwatershed surrounding the Marsh. The SWAT model was utilized to estimate streamflow in place of AnnAGNPS because the irregular shape of the Marsh. Producing streamflow results that enter the Marsh requires several subwatersheds to account for the total drainage area. Delineating around the Big Creek Marsh and accounting for all the Marsh's inlets was impractical to do with AnnAGNPS; however, the task was simpler with SWAT. For this reason the SWAT model was used to estimate streamflow entering the Marsh. The SWAT modelling of the Big Creek Watershed streamflow is outlined in the *Big Creek Watershed Plan - support for water quantity* (ERCA, 2011a). The estimated streamflow entering the Marsh includes direct surface runoff, tile drain flow, and baseflow (groundwater flow). A summary of the monthly average streamflow entering the Marsh is outlined in Table 4-2. The values in the table are for the twenty year period from 1990 to 2009. In the WB model daily inflows are used.

Table 4-2: Monthly Average Entering the Marsh

Month	**Surface Runoff (mm)**	**Tile Drain Flow (mm)**	**Groundwater Flow (mm)**	**Total Streamflo w (mm)**
Jan	30	4	1.4	35
Feb	41	3	0.9	44
Mar	36	5	0.8	42
Apr	15	13	1.6	30
May	16	9	2.8	28
Jun	15	4	2.3	21
Jul	6	1	1.3	8
Aug	9	1	0.6	11
Sep	11	3	0.3	15
Oct	8	9	0.3	18
Nov	10	11	0.5	21
Dec	27	6	1.2	34

(Source: ERCA, 2011a)

In the Big Creek Marsh WB the control dam gate (G) would release excess water if the current day's water level was greater than the target depth (Table 4-1) and the water level in the outletting body, Lake Erie, was below the level in the Marsh. The maximum allowable outflow was determined to be a function of head difference between the Marsh and the Lake (Δh). Equation 4.3 outlines the relationship between head difference and the maximum potential daily gate release rate. In the daily WB calculations the volume released from the dam would be the lesser of 1) the difference between the actual storage and the target storage or 2) the maximum allowable outflow as outlined in equation 4.3. Additionally, to more accurately emulate operation, a tolerance surplus of 5 percent was required before the water budget model would release excess water (assuming appropriate Lake water levels). The data used to estimate the maximum daily potential release rate (Equation 4.3) is outlined in the appendix.

$$Q_{max} = 785{,}399 * \Delta h^{1.44} \ \{\text{were } \Delta h \text{ is in m \& } Q_{max} \text{ is in m}^3/\text{day}\} \qquad \text{Equation 4-3}$$

4.3.4 Pumping Inflow and Outflow

In the general formulation of the WB it was first assumed that pumping both in and out of the Marsh was potentially possible. Under each of the three operating phases the permissible months of normal operation and contingency operation pumping both into and out of the Marsh is explicitly outlined. Therefore if the operating phase was know for a given year the months of allowable pumping would also be known. Similar to the estimates of the control dam released water (G), the pumping (P_{in}) from Lake Erie into the Marsh and the pumping out of the Marsh (P_{out}) were presumed to be a function of Marsh's water level, Lake Erie's water level, and the season of the wetland operation.

Pumping into the Marsh (P_{in}) is part of the normal operations under all three phases, as outlined by the PTTW (Ducks Unlimited Canada, 2007). Pumping out of the Marsh (P_{out}) in the same permit was outlined as a contingency operation for the hemi and overgrown phase, and as normal operations under the open water phase. The pumping operations for all three plans are outlined in the appendix. Both the inflow and outflow allowable pumping months are a function of the growing phase. The outflow pumping has a prescribed minimum depth of water required, whereas the inflow pumping has only a prescribed target water level. The maximum daily pumping value outflow rate is the same as inflow, but there was not a restriction on maximum annual outflow.

Limited data of recorded Marsh operations is available for the model validation, calibration, or even verification. However, three years of recorded pumping data was available for the WB modelling study. The recorded pumping data for 2006, 2007, and 2008 is summarized in Table 4-3.

Table 4-3: Recorded Pumping Data (m^3)

Month	Day of the Month	2006	2007	2008
August	1		54,504	
	2		61,771	
	3		50,870	
	4		87,206	
	5		87,206	
	6		87,206	
	7		87,206	
	8		87,206	
	9		55,158	
	10			
	11			
	12			
	13			
	14			
	15			
	16			
	17			
	18			
	19			
	20			
	21			25,435
	22			87,206
	23			87,206
	24			87,206
	25	43,603		87,206
	26	43,603		87,206
	27	43,603		87,206
	28	43,603		
	29			87,206
	30			87,206
	31			87,206
September	1			87,206
	2	43,603		87,206
	3	43,603		29,069
	4			
	5			
	6			
	7	43,603		
	8	43,603		
Annual Sum		348,826	658,336	1,013,774
Pumping Days		8	9	13

Similar to the control gate outflow (G), there are several routines that were implemented into controlling the water pumped into Big Creek Marsh. Following the prescription of the Big Creek PTTW (Ducks Unlimited Canada, 2007), water pumped from Lake Erie into the wetland could only occur in select months being a function of growing phase (refer to appendix). Additional constraints from the same permit were also employed on the volume of water pumped into the marsh; the maximum total inflow for a single day was 88370 m^3 and no more inflow than 5302400 m^3 in a calendar year. To eliminate unrealistic pumping of minor volumes of water into the marsh, a 5 % water deficit below the target level was required before the WB model would pump. In addition to the 5 % deficit requirement, if pumping was initiated it was required that the model pump for two days. The model also checked to ensure that pumping and sizable storm events would not occur within the same two day period. This final logic requirement was assumed reasonable as operators would not pump water into the Marsh if a large storm event was predicted in the following two days.

In the recorded pumping data (Table 4-3) only pumping into Big Creek Marsh is delineated. Data for pumping out of the Marsh into Lake Erie was not provided. From the non-existence of this data it is therefore assumed that the operations for these three years did not include any pumping out or outflow pumping was not recorded. If pumping out of the Marsh did not occur the likely operating scenario in 2006, 2007, and 2008 years was not the open water phase. However, since a limit on the volume of the outflow pumping was not prescribed in the PTTW (Ducks Unlimited Canada, 2007) if pumping out did occur it simply may not be mandatory to record.

4.3.5 Evapotranspiration

Evapotranspiration (ET), a fundamental component of the WB, was accounted for in the hydrological balance using the FAO 56 Penman-Monteith method. The potential evapotranspiration (PET) rate was used to estimate the outflow of this WB component. In chapter 2 a review of ET in wetland WB was outlined. Conflicting results were presented as to whether PET was a reasonable estimate for the actual evapotranspiration (AET) in a wetland. A study in a humid subtropical wetland with an unlimited supply of water found that AET rates were less than PET rates (Shoemaker and Sumner, 2006). However, a study reviewing AET in a wetland in the Great Lakes Basin found that the Penman PET methodology provided a reasonable estimate of AET rates (Soucha et al., 1996). In several other studies PET was used to account for the ET outflow in a WB.

In the Big Creek Marsh WB the PET rates were determined using the REF-ET software package (University of Idaho and Allen, R., G., 2001). Data inputted into REF-ET includes daily precipitation, relative humidity, maximum temperature, minimum temperature, average sunshine hours, and geographical location. The FAO 56 Penman-Monteith reference crop (.12 m tall grass) ET values were used directly without adjustment for crop type. The average daily PET value for month is contained in Figure 4-3. Without knowledge of a reasonable crop coefficient, this assumption is presumed to be valid. To determine the daily volumetric flow rate of the ET portion of the balance the surface area of the open water in the marsh was multiplied by the corresponding depth.

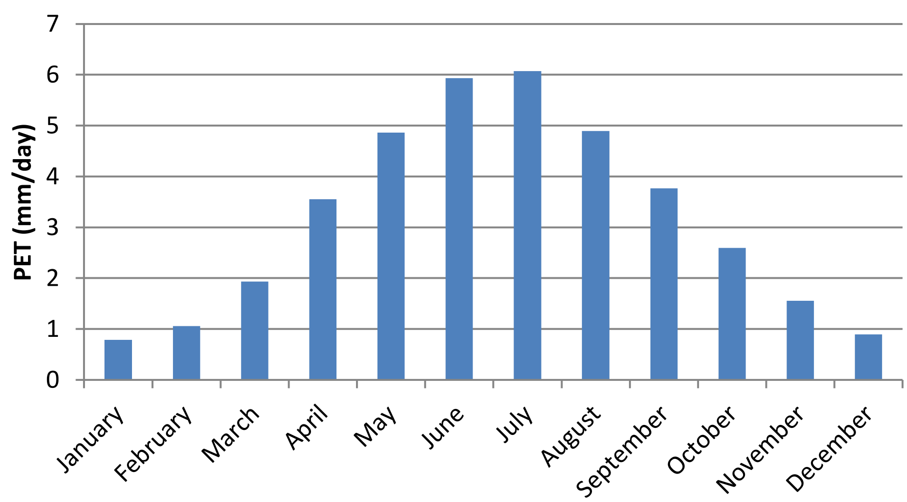

Figure 4-3: Average Monthly Precipitation (mm)

4.3.6 Inflow and Outflow Seepage

In the Marsh WB only shallow groundwater flow is considered. Groundwater flow (base flow) from streamflow is included in the SWAT model streamflow input. Deep groundwater processes are not emulated in the model. However, seepage between Big Creek Marsh and Lake Erie is approximated in the model. Seepage into (S_{in}) and out (S_{out}) of Big Creek is estimated using Equation 4-4 (Todd, 1959). In the review of wetland WB outlined in chapter 2, it was found that Darcy's Law under predicts actual groundwater seepage (Hunt et al., 1996), thus an alternative equation was utilized to estimate seepage.

$$q = \frac{K}{2l}\left(h_{higher}^2 - h_{lower}^2\right) \qquad \text{Equation 4-4}$$

In Equation 4-4 q represents the volumetric flow of seepage per unit width, K represents the hydraulic conductivity of the soil, and l represents the length of travel. The h_{higher} and the h_{lower} represent the depth of water in the higher body and the depth of water in the lower body, respectively, measured from a common datum. The datum of seepage flow interaction depth was assumed to be 172 meters above sea level (ERCA, 2011b). This constant datum of the impermeable layer in the wetland's soil stratification is not realistic (Favero et al., 2007; Hayashi and Van Der Kamp 2009), but was implemented as a reasonable representation of a normal depth. The daily Bar Point (02GH009) water levels were used as the actual Lake Erie water levels. This data was procured from the Environment Canada hydrometric data website (Environment Canada, 2010). See Section 4.3.8 for a more detailed review of the Lake Erie water level data.

The 4.5 km length of Big Creek Marsh adjacent to Lake Erie was used to determine the cross-sectional flow area, and to determine an average length of flow travel. When the head at Lake Erie was higher than the head at Big Creek, seepage was assumed to enter the control volume: if the opposite was true, seepage was assumed to leave the control volume. In the WB model seepage could only occur in one direction on any day.

4.3.7 Overflow Into and Out of the Marsh

Both flows over the control dam into and out of Lake Erie ($\mathrm{OF}_{\mathrm{out}}$ and $\mathrm{OF}_{\mathrm{in}}$) were accounted for in the WB budget model. The top elevation of the control dam is 174.90 meters above sea level. A simple logic rule dictates if flow will go over the dam and the associated quantity. The logic statement evaluates the water levels in the Marsh, Lake Erie, and then compares the same to the top of the hydraulic structure. If the water level in the Marsh is higher than the control dam, and the water level in the Lake, overflow will pour into the Lake. Conversely, if the Lake's and Marsh's water levels were reversed then it is assumed that overflow would pour into the Marsh. The water level in the Marsh was calculated daily from the WB storage volume, using Equation 4.2. Given the restricted information related to the dam overflow characteristics, estimating the quantity of water that would overtop the structure was problematic. It was simply assumed in both cases of overflow that 70 % of the volume generated from the head difference (using Equation 4.2) would travel from the elevated body of water to the lower.

4.3.8 Lake Erie Water Levels

Big Creek Marsh outlets directly to Lake Erie. The Lake's water levels directly affect the Marsh's hydrological state. The Lake effects the seepage between the two bodies, rate of flow from the control gate to the Lake, and flow overtopping the dam structure. To approximate the effect of the Lake on the Marsh the closest hydrometric water level recording station was utilized in the model. The Lake Erie at Bar Point (02GH009) continuous water level recording station levels were used as the Lake's water levels in the model. In Figure 4-4 the annual average water levels at the Bar Point station for a forty year period (1969 to 2008) are outlined.

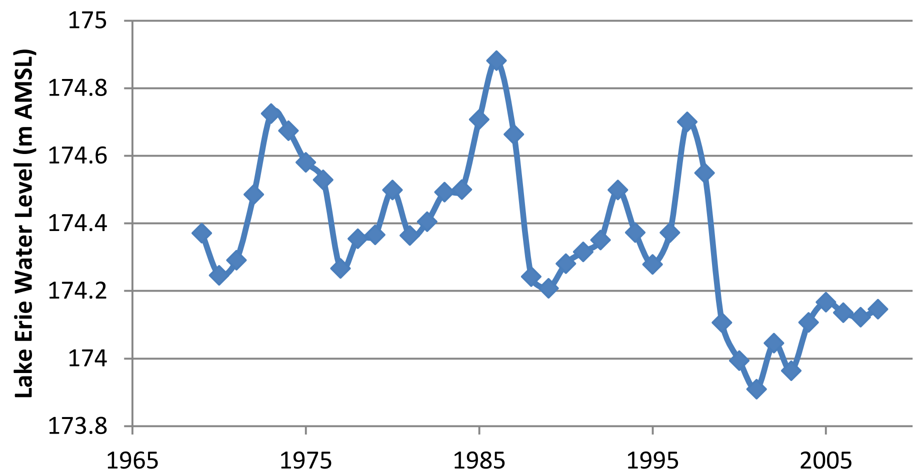

Figure 4-4: Lake Erie Average Annual Water Level (m AMSL)

Bar Point station has continuous data from 1965 to 2011. This data was procured from the Environment Canada hydrometric data website (Environment Canada, 2010). In the forty year Lake water level dataset 154 days were missing data. The missing data for those days was simply filled with an average of the day before the missing data and the day after the missing data. The average daily water level at the Bar Point station was 174.355 m AMSL, the minimum daily water level was 173.14 m AMSL, and the maximum daily value was 175.26 m AMSL. The forty year modelling period was selected in part because of the availability of the Lake's water level data.

4.4 Summary

This chapter reviewed the general structure of the Big Creek Marsh WB, explained how individual hydrological model variables were assessed, and outlined any potential perceived deficiencies. The hydrological components incorporated in the WB model include precipitation, ET, streamflow, gate controlled release, seepage inflow, seepage outflow, pumping in, pumping out, gate overflow out, and gate overflow in. The WB model follows a daily mass balance procedure, where each inflow and outflow are also calculated daily. The water level in the Marsh is estimated using a storage depth curve approximated with the wetland's bathymetry. The ET in the model is calculated using REF-ET with the FAO 56 Penman-Monteith method.

The chapter also reviewed the three potential operating scenarios suggested by the Big Creek pumping PTTW (Ducks Unlimited Canada, 2007). The three phases include the overgrown phase corresponding to a wet year, the hemi phase corresponding to an average year, and the open water phase corresponding to a dry year. The Marsh has limited recorded data of historic operations, excluding three years of pumping data from 2006 to 2008.

CHAPTER 5: AnnAGNPS MODELLING IN BIG CREEK

5.1 Introduction

The present chapter contains a brief description of AnnAGNPS input database, outlined in section 5.2. Only a select portion of the model inputs are included in this section. A summary of the AnnAGNPS model calibration and validation in the neighbouring Canard River Watershed is contained in Section 5.3. The calibration and validation only considers the hydrological components of the model. In section 5.4 the validated AnnAGNPS input database is implemented in the Big Creek Watershed simulation and the results are summarized. Figure 5-1 shows the location of the Big Creek and Canard River Watersheds in Essex County, Ontario.

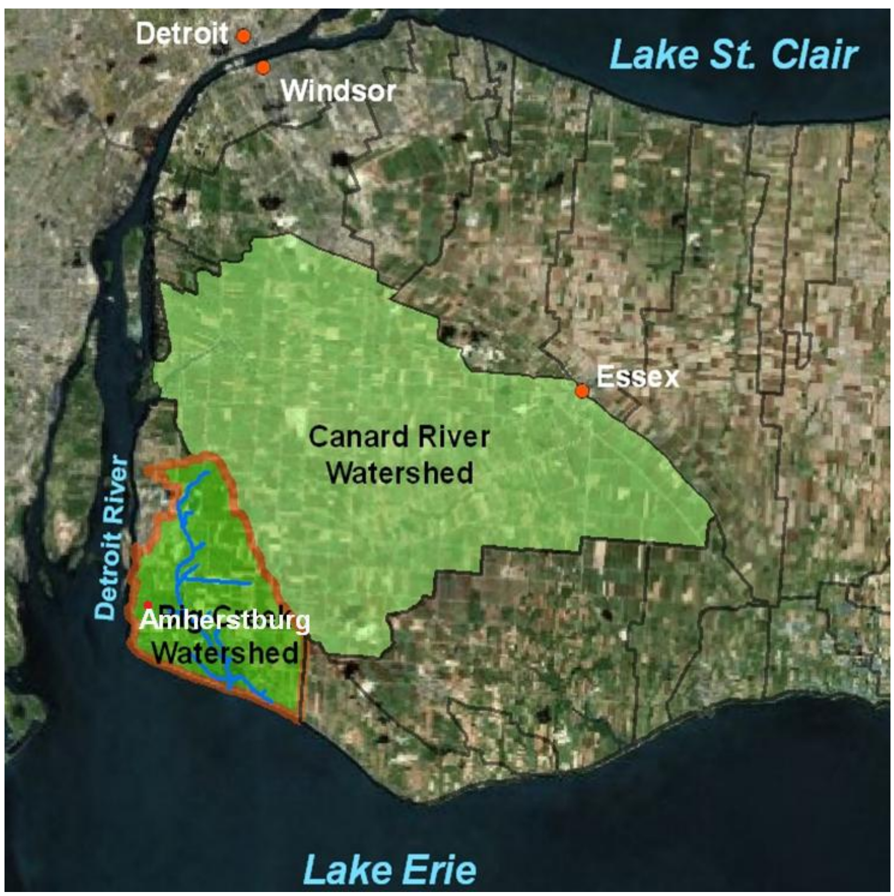

Figure 5-1: Big Creek and Canard River Watershed

5.2 AnnAGNPS Input Database

The following section contains a summary of the AnnAGNPS input database. Later in this chapter a model calibration and validation is outlined. The variables considered for the calibration were selected based upon the literature review in Chapter 2. Only the final iteration of variables modified during the calibration process will be outlined. The meteorological data will be reviewed for both the Canard River Watershed and the Big Creek Watershed.

5.2.1 Meteorological Data

Weather data is the first fundamental data input necessary to derive output results from the AnnAGNPS model with selection of appropriate data essential to emulate physical processes (Bingner et al., 2009). In the AnnAGNPS Big Creek and Canard River modelling twenty two years of historical meteorological data were utilized. The period of data used in the modelling study ranges from 1988 to 2009. The first two years of the weather data are used as an initialization period to help warm up the AnnAGNPS model simulation. The Canard River dataset spans 1990 to 2005 and the Big Creek dataset spans from 1990 to 2009. The relevant climatic data for the two watersheds were obtained from two weather stations. The Amherstburg and Windsor Airport Weather stations (Climate ID: 6130257 & 6139525) were both utilized to generate two meteorological input datasets. The meteorological data was procured from Environment Canada's Climate Data Online (Environment Canada, 2011).

The Amherstburg weather station is located within the Big Creek Watershed. The Windsor Airport weather station is located to the north of the Canard River Watershed. To implement the AnnAGNPS model a complete chronological set of climate data is required for the entire simulation period. Six daily meteorological parameters are needed for the AnnAGNPS simulation; minimum air temperature; maximum air temperature; precipitation; dew point; sky cover; and wind speed. The Amherstburg weather station only records for three of the six meteorological parameters required for modelling. The Big Creek climate data is composed of temperature and precipitation data from the Amherstburg station, but the sky cover and wind speed data is taken from the Windsor Airport station. The Canard climate data is composed entirely of data from the Windsor Airport station.

The sky cover data can be replaced with solar radiation at the ground level if available. Sky cover data for the model simulation was estimated by converting the day's weather description to a percent. A table containing the conversion is contained in the appendix. Figure 5-2 compares the annual total precipitation of the Windsor Airport weather station to the Amherstburg weather station. The annual average depth of precipitation over a period of twenty two years at the Amherstburg and Windsor Airport stations was 901 mm and 906 mm, respectively.

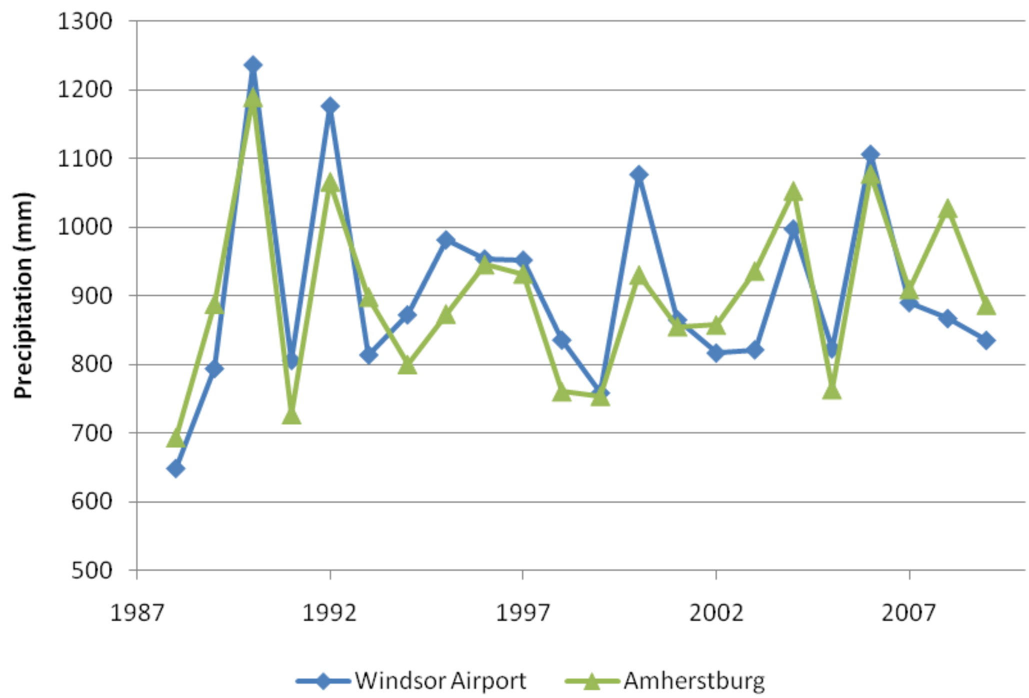

Figure 5-2: Windsor Airport and Amherstburg Weather Station Annual Precipitation

Table 5-1 outlines each month's average daily climate data. The statistics presented in the same table are for the twenty two year period of 1988-2009. The values were determined by summing all data points within the set that occurred within the subject month and then were divided by the number of days in that month and twenty two to account for the number of years.

Table 5-1: Climate Data Daily Monthly Averages

	Windsor Airport				**Amherstburg**		
	Max Temp	**Min Temp**	**Dew Point Temp**	**Wind Speed**	**Max Temp**	**Min Temp**	**Dew Point Temp**
Month	(°C)	(°C)	(°C)	(m/s)	(°C)	(°C)	(°C)
January	0.4	-6.5	-7.3	5.2	0.5	-6.4	-8.6
February	1.3	-6.2	-7.5	4.9	1.7	-6.1	-8.9
March	6.7	-2.2	-4.0	5.0	7.1	-2.3	-5.5
April	13.8	3.5	0.8	4.9	14.1	3.5	-0.9
May	20.3	9.5	6.7	4.3	20.7	9.6	4.8
June	25.9	15.4	12.5	3.8	26.3	15.6	10.7
July	28.0	17.9	14.9	3.5	28.5	17.8	13.0
August	26.9	17.2	15.4	3.1	27.3	17.3	13.5
September	22.9	12.8	11.1	3.5	23.3	13.0	9.3
October	16.0	6.8	4.1	4.2	16.1	6.8	2.9
November	8.7	1.4	0.0	4.8	8.7	1.3	-1.5
December	2.2	-4.1	-5.2	4.7	2.3	-4.2	-6.4

5.2.2 GIS Data

This section will briefly discuss the geographic information systems (GIS) datasets utilized in the Big Creek and Canard River AnnAGNPS modelling. In the AnnAGNPS software package the GIS component generally requires at least three input files for modelling; landuse, soil, and topographic. The fourth optional GIS input file is the climate station location information; this information is used to assign different climate stations to individual cells within a delineated watershed. Only one climate database was used for each watershed model, therefore the fourth GIS dataset was not required. The three GIS input files required for each watershed model was obtained from the Essex Region Conservation Authority (ERCA). Figure 5-3 and Figure 5-4 contain a graphical representation of the GIS soil land distribution data.

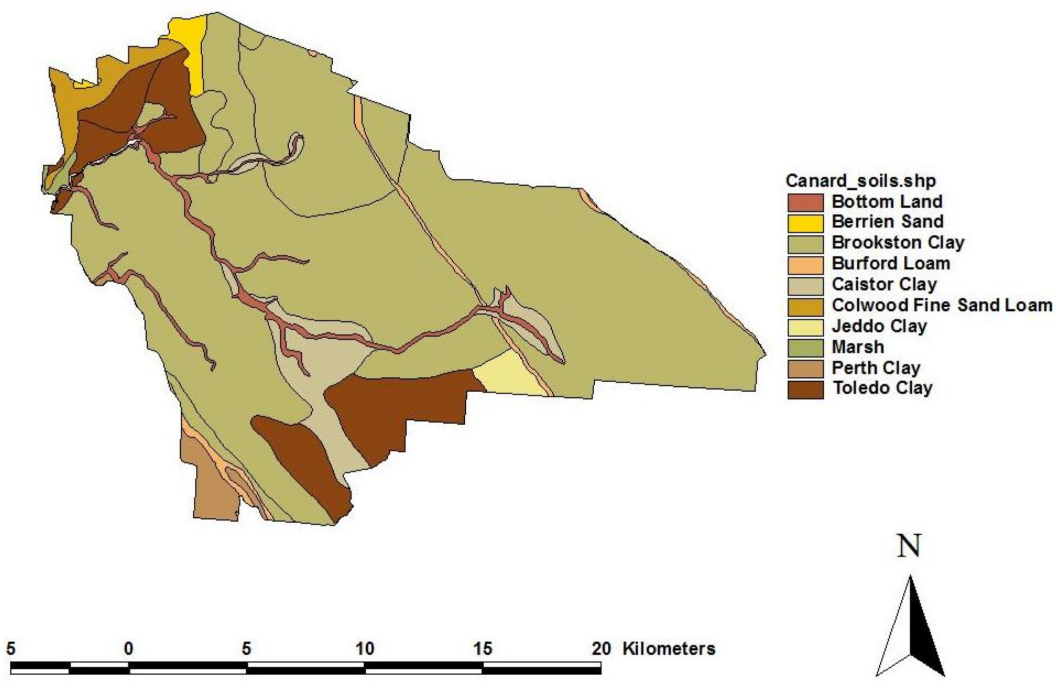

Figure 5-3: Canard River Soils

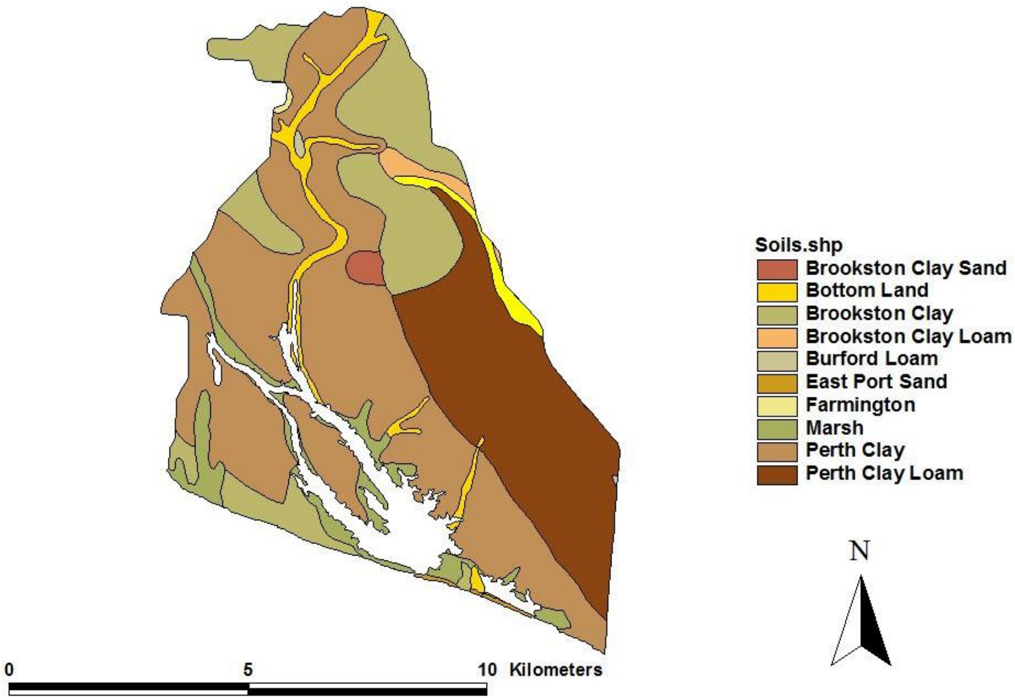

Figure 5-4: Big Creek Watershed Soils

5.2.3 AnnAGNPS Database

The following section provides a summary of the AnnAGNPS text database for the Big Creek and Canard River Watershed models. This portion of the AnnAGNPS input data is the most complex and intricate from the modeller's perspective. The major categories of data inputted into the model include soil data, tile drain data, runoff Soil Conservation Service curve numbers (SCS CN) data, non-crop data, fertilizer data, crop data, and landuse management data for outlining the temporal changes in-cells over the simulation period.

Table 5-2 contains the calibrated SCS CN data for the two watershed models. The CN numbers were originally obtained from the *Agricultural Handbook 703* (USDA, 1996) and from *Technical Release 55: Urban Hydrology for Small Watersheds* (NRCS, 1986). The original CN were modified by both subjective and objective assessment

(Nash-Sutcliffe coefficient, coefficient of determination and index of agreement), with an iterative process attempting to best match the observed streamflow data.

Table 5-2: SCS Curve Numbers

Curve Number ID:	**CN "A"**	**CN "B"**	**CN "C"**	**CN "D"**
Small Grain Straight Row Poor	69	78	83	87
Crop Residue Cover	71	78	86	87
Rangeland Poor	65	74	85	87
C1	30	55	70	75
C2	44	63	76	82
C3	53	68	78	81
C4	52	69	78	82
C5	58	72	80	84
C6	61	75	83	85
C7	74	83	88	90
Buffer	30	47	64	72
Urban Pervious	81	85	89	91
Urban Impervious	86	91	92	92
Pasture (Poor)	79	79	79	81

The values in Table 5-2 correspond to the average SCS CN (CN_2) for each soil group type A, B, C, or D. The fourteen CNs outlined in the table correspond to different landuses and vary with the landuse management practices. The average CN for each CN ID is modified during the simulation for soil moisture conditions as discussed in Chapter 3. The original CN for the alpha-numeric C1 to C7 are defined for each respective landuse type in the *Agricultural Handbook 703* (USDA, 1996). The remainder of the CN datasets were originally taken from *Technical Release 55: Urban Hydrology for Small Watersheds* (NRCS, 1986).

The fertilizer application data is fundamental for estimating nutrient runoff in AnnAGNPS simulations. Table 5-3 outlines the fertilizer application rate and composition. There are five fertilizer applications modelled in AnnAGNPS. Three of the fertilizers in the table are preliminary applications of phosphorus to the field for each of the three crops assumed to be grown in the agricultural fields.

Table 5-3: Fertilizer Application and Composition

Fertilizer Name ID:	**80-20-0**	**20-0-20**	**PSB**	**PCRN**	**PWW**
Fertilizer Application ID:	sfww1	sfww2	sfww3	sfww4	sfww5
Fertilizer Rate (kg/ha):	85	135	11	20	8
Fertilizer Depth (mm):	10	10	10	10	10
Inorganic N (wt/wt):	0.6	0.8	0	0	0
Organic N (wt/wt):	0.2	0.1	0	0	0
Inorganic P (wt/wt):	0.2	0.1	1	1	1
Organic P (wt/wt):	0	0	0	0	0
Organic Matter (wt/wt):	0	0.2	0	0	0
Consistency:	Solid	Solid	Solid	Solid	Solid

In the AnnAGNPS model the fertilizer PWW is applied in early October before the planting of winter wheat. The fertilizer 80-20-0 is applied in the middle of February to the same winter wheat crop. The PSB fertilizer is applied in the first week of June just before soybean is planted. A fertilizer with nitrogen content is not applied to fields with soybeans. The PCRN fertilizer is applied in the second week of May just before the planting of the corn crop. The 20-0-20 fertilizer is applied at the beginning of September also to the fields planted with corn.

The AnnAGNPS model's assumed planting and harvesting dates are outlined in Table 5-4. The crop rotations in the landuse management data utilized in the AnnAGNPS database follows two six year rotations and one four year rotation in accordance with the dates summarized in Table 5-4. Agricultural landuse practices and fertilizer application (Table 5-3) data were obtained from *Agronomy Guide for Field Crops – Publication 811* (OMAFRA, 2002). Additionally, specific details pertaining to agricultural landuse disturbances and manipulation were taken from a database included with the AnnAGNPS software package.

Table 5-4: Planting and Harvesting Dates

Crop	Planting Date	Harvest Date
Corn	2nd Week of May	3rd Week of September
Winter Wheat	1st Week of October	2nd Week of July
Soy Bean	1st Week June	2nd Week of October

Soil data is a fundamental component of the AnnAGNPS simulation significantly influencing the model outputs. The soil in the AnnAGNPS cells acts as storage for water and nutrients, drains quick return flow to the reaches, its moisture conditions dictate runoff rates, and is the source of sediment yield. The input data for the AnnAGNPS model comes primarily from three sources. The first source of data is the Soil Survey of Essex County (Richards et al., 1949), a report of the general soil characteristics and profiles in the Essex County region. The report includes the approximate depth for each soil profile, descriptions of the soil size distribution in each layer, and descriptions of organic matter content. Little data about phosphorus content is contained within the report; however, it is stated that in approximately 80 % of the 105 soil samples analyzed in the report the phosphorus content was less than 200 lb/acre (phosphorus deficient). The soil nitrogen content is not discussed. The second major source of data is the Soil Water Characteristics (Saxton, 2009) software tool. This tool estimates the saturated hydraulic conductivity, base saturation, and wilting point when the soil size type distribution and organic matter content is inputted.

The final source of input data for the model's soil characteristics were the default values in the AnnAGNPS model. Accurate nutrient content for the soil was not readily available; therefore the default value was used for the first layer. A decreased or equal nutrient content would then be used for the remaining lower layers.

5.3 AnnAGNPS Model Calibration and Validation

The non point source (NPS) and water budget (WB) modelling in the Big Creek Watershed has limited observed data to calibrate, validate or even verify the AnnAGNPS predictions. Less than eight complete months of recorded streamflow data are available within the Big Creek Watershed. This eight month dataset only provides a restricted verification at best of the model predictions. However, the Canard River Watershed has a streamflow gauging station with observed daily streamflow values for close to a 35 year period. The Canard River Watershed is an adjacent watershed sharing Big Creek's northern and eastern boundaries. Because of the close proximity, available observed streamflow data and similar characteristics the Canard River Watershed, the AnnAGNPS input database can be calibrated and validated in this drainage area. The validated data set can then be used in the Big Creek Watershed AnnAGNPS model.

5.3.1 Canard River Watershed Calibration

This section discusses the AnnAGNPS model calibration and validation in the Canard River Watershed. A sixteen year period was selected for the model calibration and validation, eight years for each model review process. The streamflow data for the Canard River gauging station sub-watershed was obtained from the Environment Canada - Hydrometric Data website (Environment Canada, 2010). The Lukerville gauging station is located within the Canard River Watershed. The AnnAGNPS delineation of the subwatershed draining to the gauging station is outlined in Figure 5-5. In the same figure the subwatershed is overlaid on top of the Canard River Watershed. The drainage area of the subwatershed was delineated in AnnAGNPS as 170 km^2 whereas the drainage area on the of the Lukerville gauging station on the Environment Canada - Hydrometric Data website (Environment Canada, 2010) was stated to be 159 km^2. To equally compare both the AnnAGNPS predicted dataset and the recorded streamflow dataset, both were converted to depths of streamflow.

The first eight year period of streamflow data, 1990 to 1997, will be utilized in model calibration. The second eight year period, 1998-2005, will be utilized in the model validation. The calibration and validation datasets are analysed both with objective and subjective reviews. The subjective assessment in the calibration process included visual inspections of hydrographs and comparison on averages. The objective assessment utilized mathematical measures to compare how well the model predictions match the observed data.

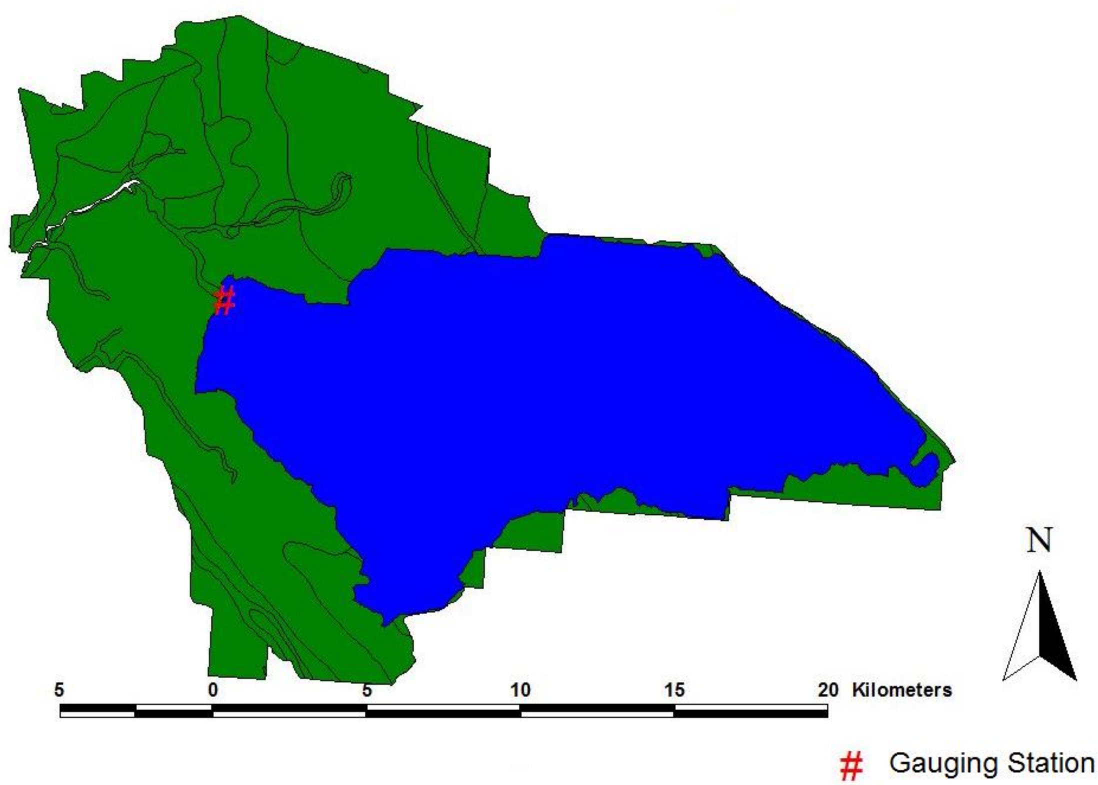

Figure 5-5: Canard River Gauging Station Subwatershed

The variables considered for the calibration were determined from the literature review of AnnAGNPS outlined in Chapter 2. The majority of reviewed studies considered only the CN for calibration of AnnAGNPS streamflow. The AnnAGNPS model's runoff was found to be sensitive to two other model parameters, wilting point and field capacity (Jayasuriya, 2007). The wilting point and field capacity are directly related to the water storage capacity of the soil. These two variables dictate the maximum and minimum potential CN; therefore, the variables considered in the model calibration were CN, wilting point, and field capacity.

Three objective assessments were implemented to investigate the match of the predicted and observed data. The three assessments considered are the Nash-Sutcliffe efficiency, coefficient of determination, and index of agreement. The three aforementioned efficiency criteria are commonly utilized in the evaluation of hydrological modelling (Krause et al., 2005). The Nash-Sutcliffe efficiency is a coefficient representing how closely the modelled values match the observed values. The Nash-Sutcliffe efficiency is outlined in Equation 5-1.

$$E = 1 - \frac{\sum_{i=1}^{n}(O_i - P_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2} \qquad \text{Equation 5-1}$$

The coefficient of determination is defined as the squared value of the coefficient of correlation and describes how much the observed dispersion is explained by the prediction (Rahman, 2007). A simpler definition of the coefficient of determination is a measure of how consistently do modelled values compared to observed values match a line of best fit. The coefficient of determination is outlined in Equation 5-2.

$$r^2 = \left(\frac{\sum_{i=1}^{n}(O_i - \bar{O})(P_i - \bar{P})}{\sqrt{\sum_{i=1}^{n}(O_i - \bar{O})^2}\sqrt{\sum_{i=1}^{n}(P_i - \bar{P})^2}} \right)^2 \qquad \text{Equation 5-2}$$

The index of agreement represents the ratio mean squared error to the potential error. The formula for the index of agreement is outlined in Equation 5-3. The index of agreement is insensitive to low flow conditions similar to the Nash-Sutcliffe efficiency (Krause et al., 2005). Each of the three objective assessments would provide a limited view of the model’s efficiency of predictions; therefore, the three were utilized to provide a broader scope.

$$d = 1 - \frac{\sum_{i=1}^{n}(O_i - P_i)^2}{\sum_{i=1}^{n}(|P_i - \bar{O}| + |O_i - \bar{O}|)^2} \qquad \text{Equation 5-3}$$

In, Equation 5-1, Equation 5-2, and Equation 5-3 the variables O_i and P_i represent the observed and predicted (modelled) data point i, respectively. The variables $\bar{O}$ and $\bar{P}$ represents the average of the observed and predicted datasets. The values for the Nash-Sutcliffe efficiency range from 1 to - ∞, where 1 represents a prefect fit. The values for the coefficient of determination range from 1 to 0, where 1 represents a perfect fit and 0 represents no correlation between the datasets. The index of agreement range is similar to the coefficient of determination, ranging 1 to 0, where 1 represents a perfect fit and 0 represents no correlation. Table 5-5 contains a conversion chart for the efficiency numerical value to a descriptive text classification. The table was determined from a review of the available literature. The conversion in the table only corresponds to the Nash-Sutcliffe efficiency and the coefficient of determination.

Table 5-5: Classifications of Efficiencies

r^2 or E - Range	Class
>0.90	Excellent
0.89-0.75	Very Good
0.74-0.50	Good
0.49-0.25	Fair
0.24-0.00	Poor
<0.00	Unsatisfactory

(Source: Moriasi et al., 2007, Parajuli et al., 2008)

In the calibration processes larger timescale were first assessed to review model efficiency. The calibration considered the annual, seasonal, monthly and daily streamflow from 1990 to 1997. In Figure 5-6 a comparison of the predicted and observed annual total streamflow in the calibration period is outlined. The inverted axis of the same figure outlines the total annual precipitation. The average annual observed streamflow of 351 mm was slightly over-predicted in AnnAGNPS as the average annual predicted streamflow was 362 mm. The average annual over-prediction was only 3 % in the calibration phase. The trend of annual over-prediction only occurred in half of the years in the calibration period. The next step of the calibration process considered the monthly streamflows, Figure 5-7.

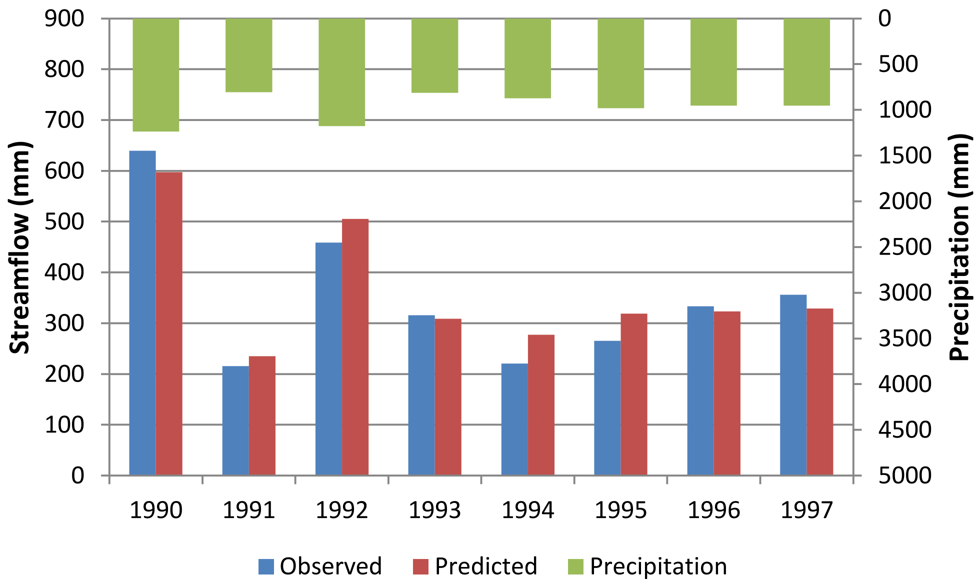

Figure 5-6: Calibration Period Predicted and Observed Annual Streamflow

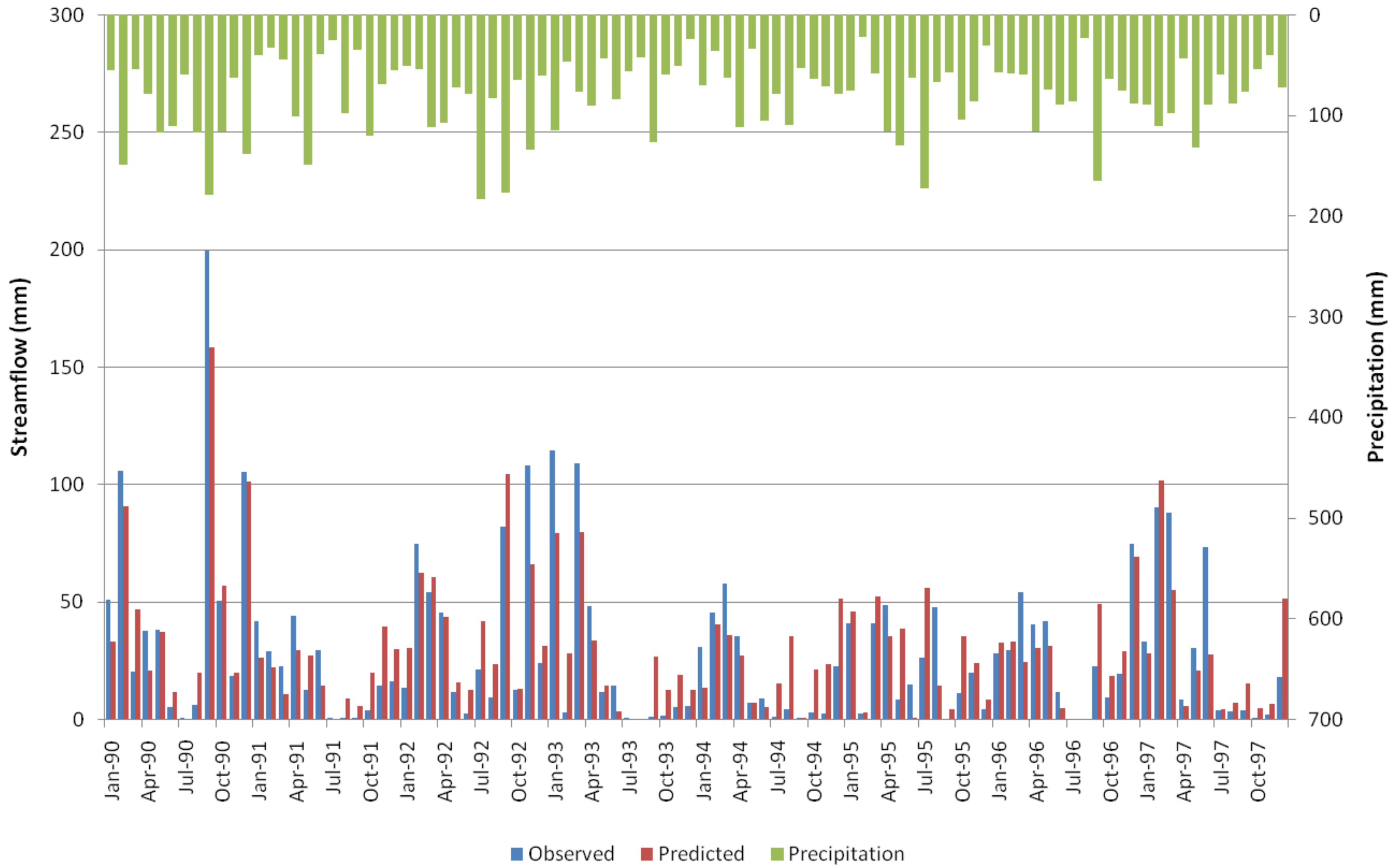

Figure 5-7: Calibration Period Predicted and Observed Monthly Streamflow

The data presented in Figure 5-6 and Figure 5-7 contains the final iteration results for the calibration process. In the annual calibration entire CN ID sets (see Table 5-2) were modified by altering all the hydrological soil groups in each set. In the monthly calibration process wilting point, field capacity, and individual hydrological soil group CNs were modified. The subwatershed in the Canard River Watershed draining to the Lukerville gauging station has a majority of clay type soils, with over 85 percent of the modelled land cover being composed of soils categorized as hydrological soil group D. In the calibration process streamflow was found to be very sensitive to the CNs associated to soil group type D.

In the monthly streamflow comparison there are several months with high streamflow where AnnAGNPS under-predicts the observed value. Nine of the 96 months in the calibration period were over-predicted by a minimum of 20 mm. The six months that had the greatest over-predictions were June 1997, November 1992, September 1990, January 1993, August 1995, and March 1993. June 1997 has a relatively high rainfall (90 mm) but this was not modelled with high streamflow. Similarly, AnnAGNPS over-predicted several months. Ten of the 96 months were over-predicted by a minimum of 20 mm. Despite these discrepancies 45 of the 96 months fell within a 10 mm prediction of the observed value.

In the final assessment of the calibration period the three objective assessment efficiency criteria discussed earlier were also reviewed. In Table 5-6 the daily, monthly, and annual model efficiencies for the calibration period are outlined. The AnnAGNPS predictions were generally excellent considering the annum basis. The monthly AnnAGNPS predictions were between good and very good. The daily AnnAGNPS predictions were generally fair. The efficiency analysis indicates that the AnnAGNPS predictions have a decreasing match with the observed data as the time period considered becomes finer.

Table 5-6: Calibration Period Streamflow Efficiencies

	Annual	Monthly	Daily
Nash-Sutcliffe Efficiency	0.919	0.746	0.328
Coefficient of Determination	0.935	0.751	0.456
Index of Agreement	0.977	0.918	0.812

5.3.2 Canard River Watershed Validation

The following section further investigates the reliability of the AnnAGNPS model’s streamflow prediction. The validation period selected for the AnnAGNPS model’s prediction was the eight years following the calibration period (1998-2005). In the validation period a similar set, as used in the calibration, of analytical metrics were calculated to investigate the adequacy of the model’s predictions. An objective review of the total annual streamflows was first considered in the model validation. In Figure 5-8, the annual predicted streamflow and observed streamflow are compared.

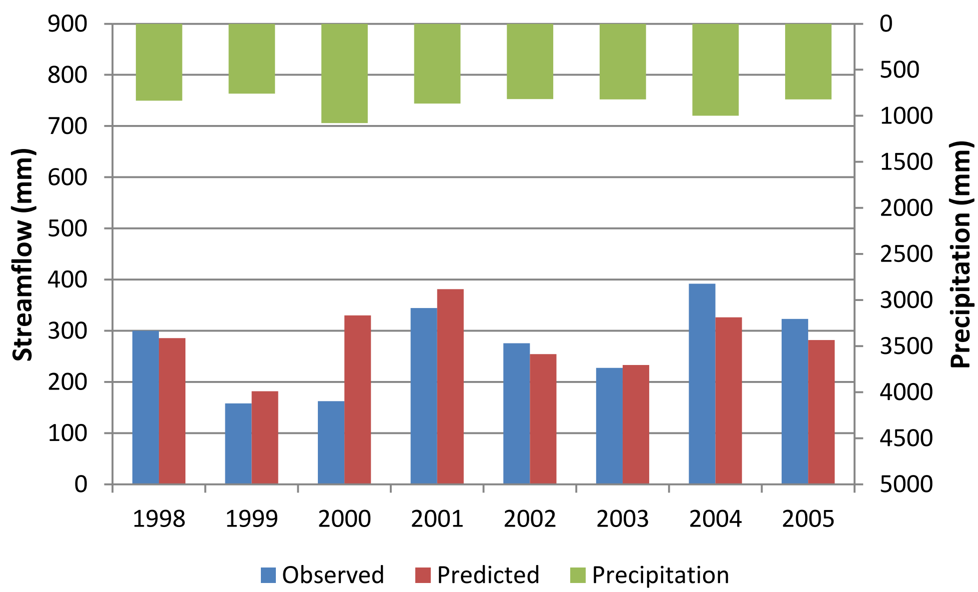

Figure 5-8: Validation Period Predicted and Observed Annual Streamflow

In the comparison of the annual streamflow in the validation period a clear discrepancy was found between the predicted and observed dataset in the year 2000. In the validation period the year 2000 had the highest annual precipitation of any year. Contrary to the high precipitation in the year 2000 the lowest annual recorded streamflow occurred in the same year. The streamflow prediction from AnnAGNPS in the year 2000 had the second highest streamflow. This discrepancy between the two recorded datasets, precipitation and streamflow, is difficult to explain but is evident in the observed and predicted streamflow. In the observed streamflow dataset for the year 2000, there were no abnormal flags on the data to indicate an issue with the records. A monthly comparison of monthly observed and predicted streamflow is outlined in Figure 5-9.

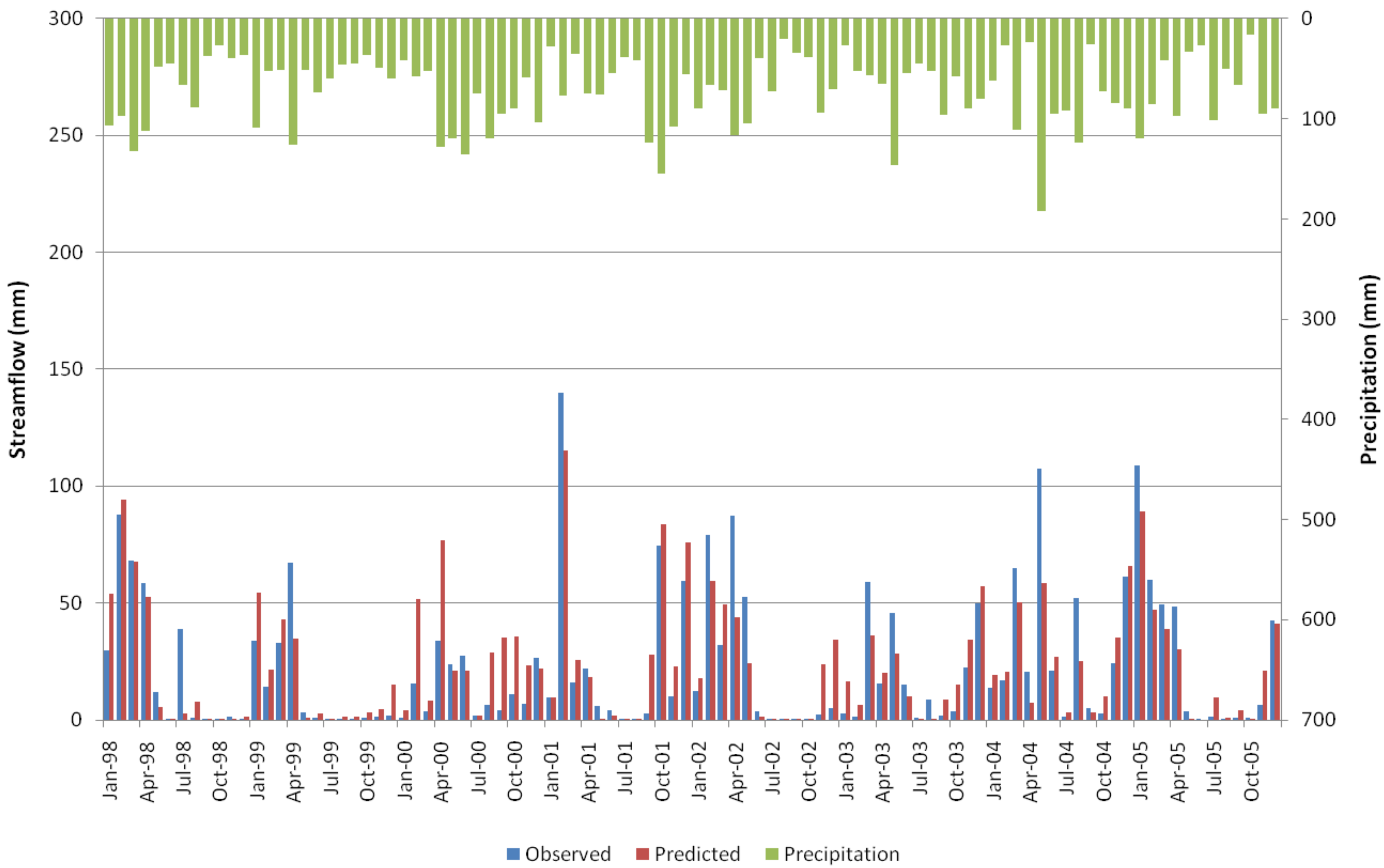

Figure 5-9: Validation Period Predicted and Observed Monthly Streamflow

In the annual analysis of the validation period streamflow datasets the AnnAGNPS outputs were slightly over-predicted, predicting an annual average of 284 mm compared to the observed streamflow of 273 mm. Four of the eight years in the validation period were under-predicted and four were over-predicted. If the outlier data year of 2000 is removed from the comparison, AnnAGNPS would under-predict the total average annual stream by 11 mm, 278 mm to 289 mm for the seven year period. A summary of the annual streamflow predictions, observed streamflow, and percent difference between the two dataset is contained in the appendix.

In the monthly assessment of the validation period ten of the 96 months were under-predicted by a minimum of 20 mm. Only eight months in the validation period were over-predicted by a minimum of 20 mm. However, five of the same eight months over-predicted were in 2000: in specific, the months of February, April, August, September, and October. Sixty one of the 96 months predicted streamflows fell within a 10 mm tolerance of the observed value.

In the assessment of the validation period the three efficiency criteria considered in the calibration were also utilized to investigate the data. In Table 5-7 the daily, monthly, and annual model efficiencies for the validation period are outlined. Opposite to the calibration period the annual predictions in the validation period generated the worst efficiency values. The validation Nash-Sutcliffe efficiency and the coefficient of determination annual values were in the fair range, whereas the same values in the calibration period were excellent. This significant reduction in the model's output matching the observed data on the annual scale is largely attributed to the discrepancy between the predicted and observed streamflow in 2000. A potential cause of this discrepancy could be an issue with the recorded streamflow data, recorded precipitation data or both datasets. If the single data point for the year 2000 was removed from the annual efficiency analysis, the Nash-Sutcliffe efficiency would increase to 0.759 (very good), the coefficient of determination would increase to 0.785 (very good) and the index of agreement would increase to 0.925.

Table 5-7: Validation Period Streamflow Efficiencies

	Annual	Monthly	Daily
Nash-Sutcliffe Efficiency	0.263	0.724	0.317
Coefficient of Determination	0.315	0.725	0.429
Index of Agreement	0.746	0.911	0.812

In Table 5-7 the trend of increasing the temporal resolution producing predictions that have a decreasing match with the observed data was not apparent in the validation dataset as was observed in the calibration dataset (Table 5-6). Including the discrepancy in 2000 the AnnAGNPS predictions were generally good considering the monthly basis. The AnnAGNPS daily validation prediction efficiencies were slightly lower than the daily calibration predictions. The daily validation predictions were still within the fair range.

In Figure 5-10 the monthly average streamflows for both the observed and predicted datasets are plotted. The averages in the figure are over the sixteen year study period to better reflect the actual trends of the drainage area. The model predicted monthly streamflow averages generally matched the months of January and February, under-predicted the spring flows in March to June, and over-predicted the remainder of the months. From a visual inspection the pattern of peaks, valleys, and relative changes in the average monthly streamflows was matched with the AnnAGNPS predictions.

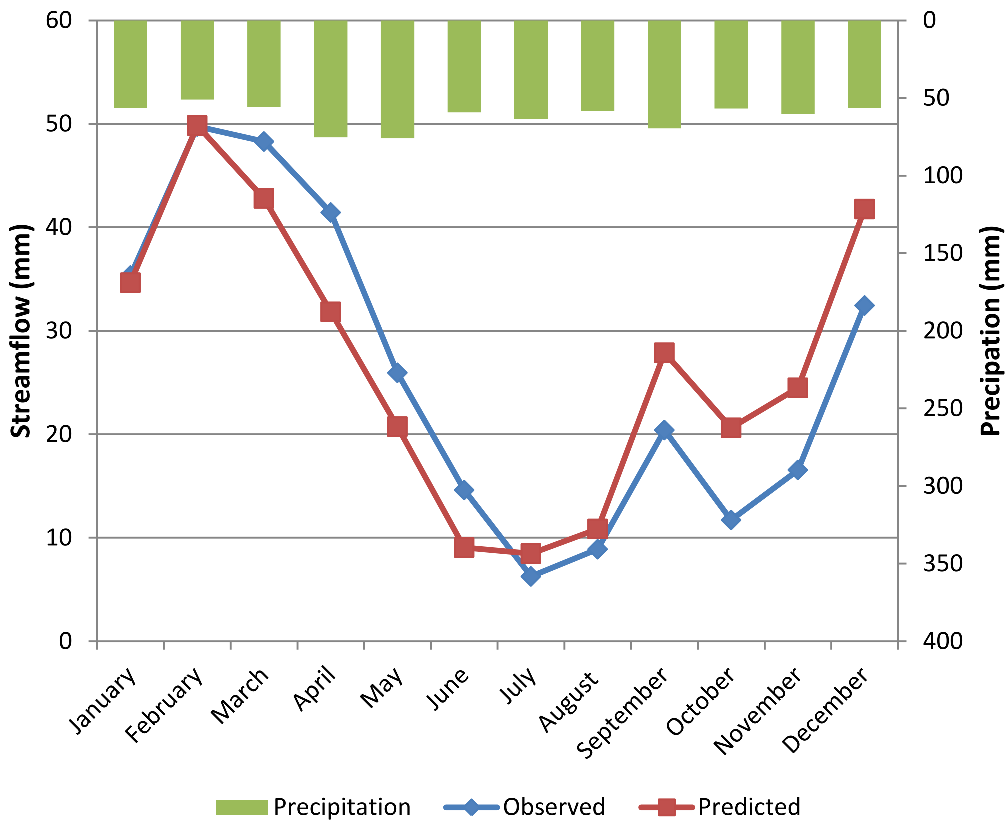

Figure 5-10: Monthly Streamflow Trends

5.4 Big Creek AnnAGNPS Results

The following section outlines the AnnAGNPS model predictions for the Big Creek Watershed. The first portion of this section further investigates the AnnAGNPS model's adequacy in the Essex Region with an eight month period of recorded streamflow from a subwatershed within the greater Big Creek Watershed.

5.4.1 Big Creek Subwatershed Verification

To further verify the AnnAGNPS model and its input database, an additional set of limited streamflow data for the watershed was utilized to investigate the model's predictions. A temporary gauging station recorded streamflow data for an incomplete eight month period from April 23, 2009 to November 19, 2009. The data was recorded continuously on an hourly basis and was converted to an average daily flow depth for comparison with AnnAGNPS predictions. There is a break in the continuous monitoring of streamflow data from July 28, 2009 to August 4, 2009. The temporary gauging station is located at the outlet of the subwatershed outlined in Figure 5-11. The data from the gauging station was the only one of three successful streamflow monitoring stations utilized in the *Big Creek Watershed Plan - support for water quantity, drainage and erosion* (ERCA, 2011a). The gauging station was located in the upstream part of the watershed at the intersection of County Road 20.

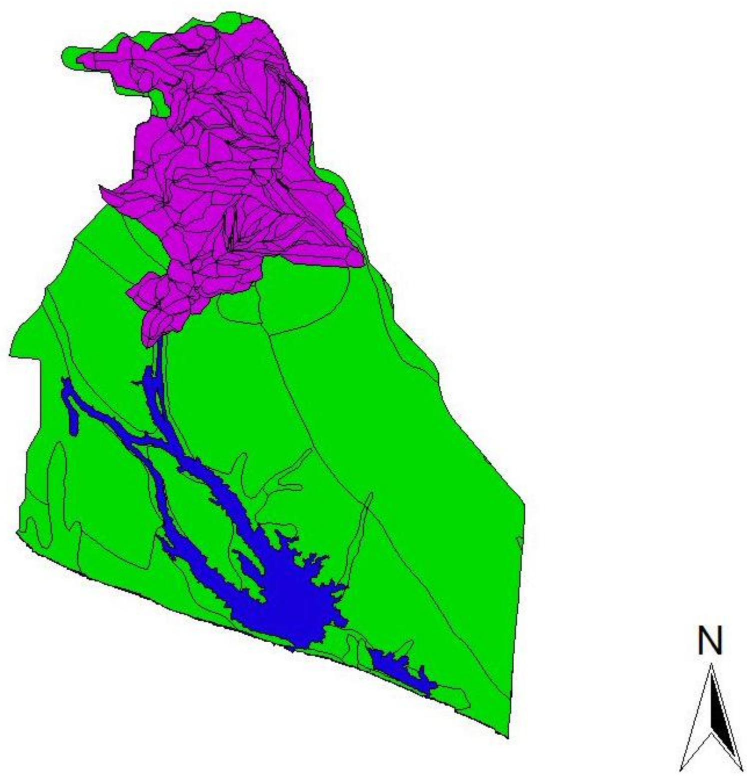

Figure 5-11: Big Creek Gauging Station Subwatershed

In Figure 5-12 the observed monthly streamflow depths are compared to the predicted depth of streamflow. An inverted hyetograph of the precipitation from the Amherstburg weather station is included in the same figure. The data presented in the figure illustrates that the AnnAGNPS predictions do not match well with the observed data. The months with incomplete data are still included in Figure 5-12. The observed streamflow dataset for the months of April, July, August, and November are composed of an incomplete number of calendar days. The same days were removed from the AnnAGNPS dataset to match the observed data. The average monthly observed streamflow was 2.4 mm; whereas, the average predicted monthly streamflow was 1.6 mm. The low streamflow is a function of the months included in the analysis, months that generally have low streamflow (see Figure 5-10), and that the month with higher streamflow, April, only has one week of data included.

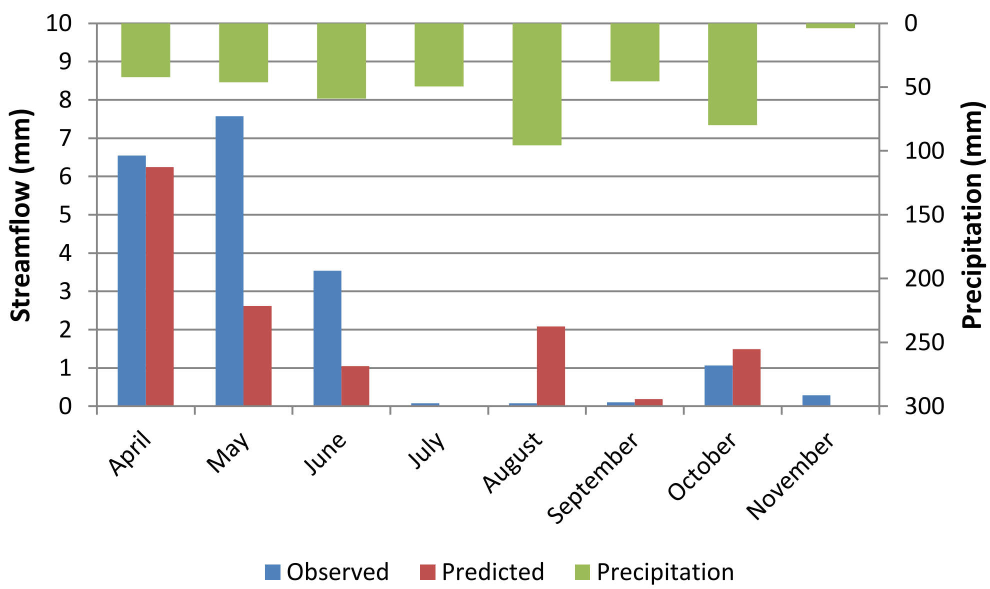

Figure 5-12: Big Creek Gauging Station Monthly Streamflow

To provide an objective review of the predicted data's match with the observed streamflow the Nash-Sutcliffe efficiency, coefficient of determination, and index of agreement were calculated with the incomplete eight months of data for the subwatershed. The daily and monthly values for each efficiency criteria are outlined in Table 5-8. Only the complete days of data were considered when calculating the values in the same table. The results of the verification period in the Big Creek gauging station subwatershed had significantly lower efficiency values than in the Canard River validation and calibration results.

Table 5-8: Big Creek Gauging Station Streamflow Efficiencies

	Monthly	Daily
Nash-Sutcliffe Efficiency	0.480	0.295
Coefficient of Determination	0.542	0.133
Index of Agreement	0.798	0.558

In the Big Creek gauging station subwatershed the monthly efficiency values were generally fair to good. The daily efficiencies were poor to fair. A possible explanation for the discrepancy was that the calibrated dataset did not include two of the major soil types in the Big Creek gauging station subwatershed: Bottom Land and Brookston Clay Sand. The discrepancy could also be attributed to the quality of the observed streamflow data. The data utilized in the Canard River calibration was from a permanent long term gauging station; whereas, the gauging station utilized to record the streamflow data for the Big Creek subwatershed was a temporary station. The break in recorded streamflow data from July 28 to August 4 is unlikely the cause of the lower values for the efficiency criteria, as only a minimal amount of precipitation, 8 mm, was recorded during this period.

5.4.2 Water Results

The following section outlines the Big Creek Watershed's AnnAGNPS model's hydrologic results. The results consider the entire watershed's drainage area that outlet to the Lake Erie. The total drainage area of the watershed estimated in the AnnAGNPS delineation was approximately 64.4km^2. The average annual effective precipitation in the Big Creek Watershed over the twenty year study period, 1990 to 2009, was 914 mm. In Figure 5-13 the annual average water budget (WB) for the twenty year study period is outlined.

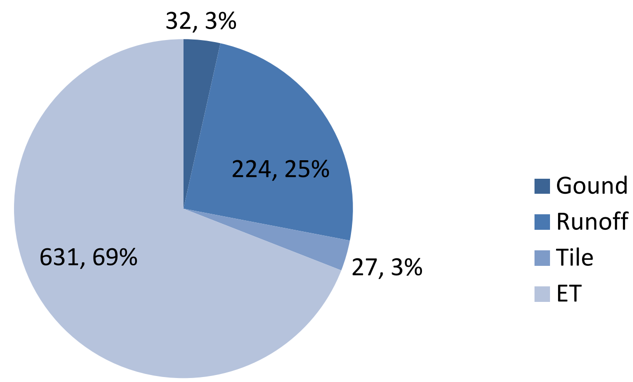

Figure 5-13: Annual Water Budget Results (mm)

The largest component of the WB is the evapotranspiration (ET) representing nearing 69 % (631 mm) of the total WB. The second largest component of the WB was direct surface runoff representing 25 % (224 mm) of the total WB. The tile drain flow represented 3 % of the total WB at 27 mm. The direct runoff and the tile drain flow represent the quick return flow in AnnAGNPS that accounts for the majority of the model's streamflow. The final component of the WB was groundwater infiltration representing 3 % of the total budget at 32 mm. In the simulation results the water yield and water loadings were not equivalent. The yield refers to the amount of a material being generated in a cell. The loading refers to the amount of material ultimately leaving the watershed though the final outlet. The outlet's water yield in the simulation was less than the water loading at the outlet, 258 mm compared to 264 mm. However, the water yield and loading from each cell was equivalent.

The annual average water loading spatial distribution is outlined in Figure 5-14. The water yield refers to both the direct surface runoff and the quick return subsurface flow (tile drain flow). The AnnAGNPS model is also capable of simulating water load contributing to streamflow from alternative sources including point sources, ponds, and irrigation. All three were not considered in the Big Creek Watershed model.

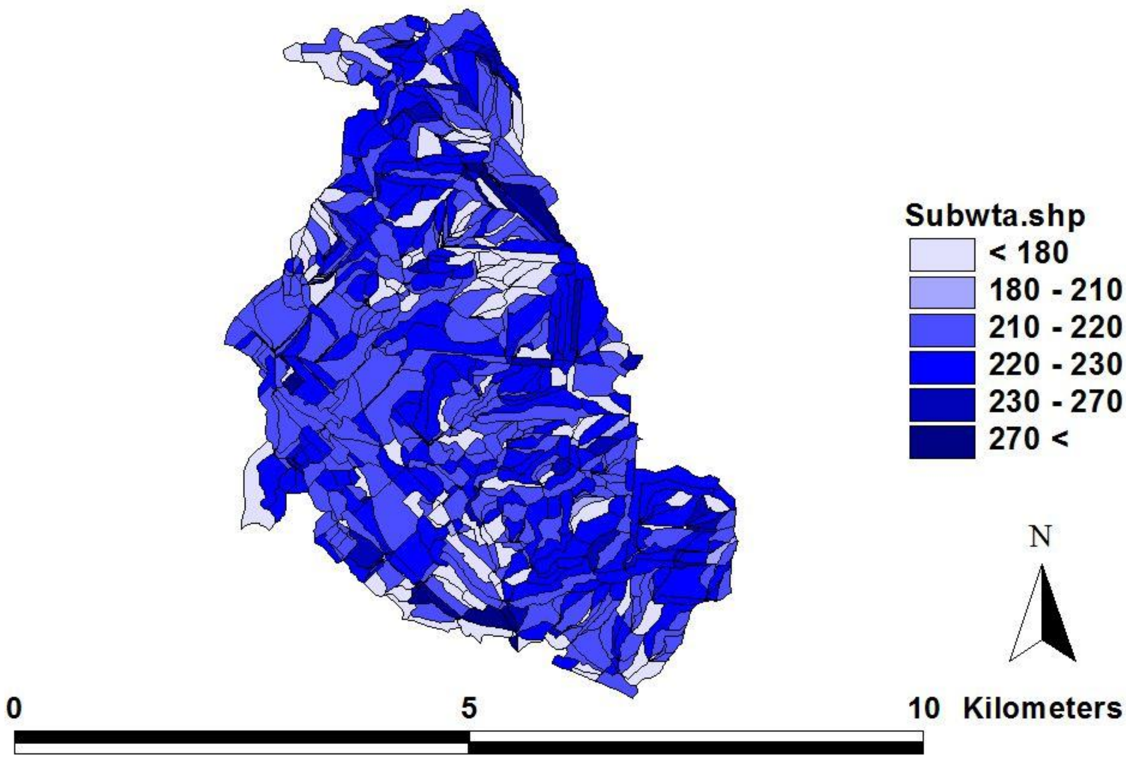

Figure 5-14: Spatial Distribution of Average Annual Water Loading (mm/yr)

The annual average water loading for the simulation period ranged from 81 mm to 914 mm. The landuse cover type producing the extremely low runoff rates were row hedges. Twelve cells were classified as openwater in the watershed. These twelve cells had a total direct runoff equal to precipitation (914 mm). The agricultural landuse types generally produced annual average loadings ranging from 180 mm to 270 mm.

The summary statistics for the annual average direct runoff of the 660 cells that the Big Creek was delineated into are outlined in Table 5-9. The same table differentiates the tile drain runoff, direct surface runoff and total runoff statistics. Figure 5-14 only considers total direct runoff. The statistics presented in Table 5-9 include the average, standard deviation (SD), coefficient of variation (CV), maximum value (Max), and minimum value (Min). The standard deviation is a measure of the variation each set has from its mean. A high standard deviation indicates a large spread of data. The coefficient of variation is a normalized standard deviation that has been divided by the subject dataset's mean. With this normalized statistic, comparisons of the variation between datasets can be more fairly contrasted. The maximum and minimum values simply represent the largest and smallest quantity in each respective set.

Table 5-9: Annual Average Direct Runoff (mm)

	Tile Drain Flow	Direct Surface Runoff	Total Runoff
Average	25.9	198.5	224.4
SD	15.8	99.6	97.3
CV	0.6	0.5	0.4
Max	100.3	914.3	914.3
Min	0.0	28.3	28.3

In Figure 5-15, daily flow rates from the modelled watershed's outlet are outlined with an inverted hyetograph of the model input precipitation data. The units for streamflow are presented as mm/day. The daily AnnAGNPS results are initially summarized in a mass per day rate. The maximum and minimum modelled daily flows were 46.5 mm and 0 mm, respectively. The average daily flow rate over the twenty year study period is 0.70 mm. In the twenty year simulation period 2103 of the 7305 days had a modelled flow of 0 mm.

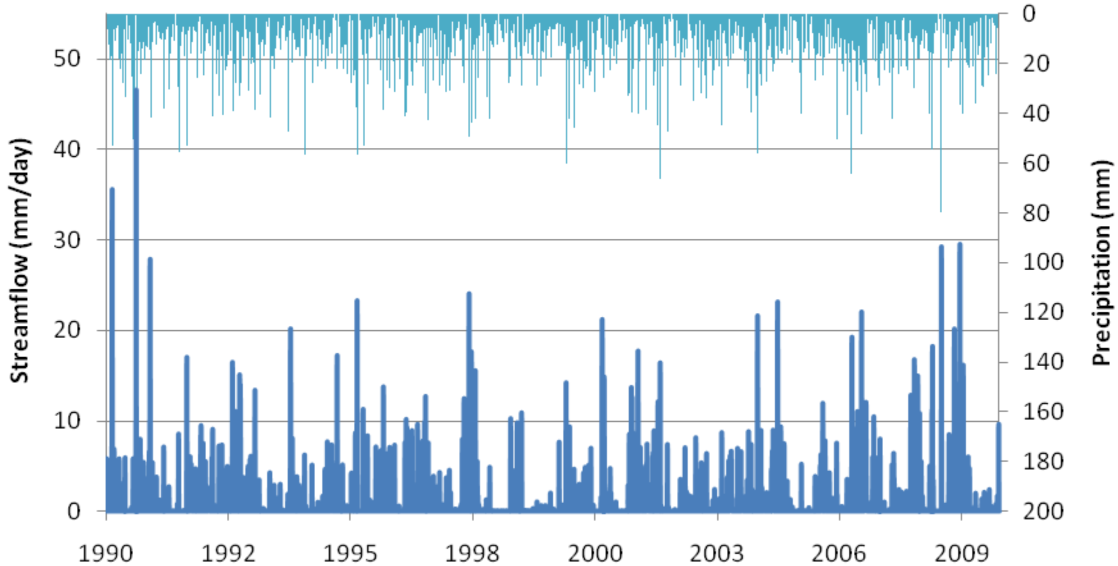

Figure 5-15: Daily Streamflow

The average monthly streamflow estimated at the watershed outlet is outlined in Table 5-10. The average annual streamflow was found to be 264 mm over the study period. In the AnnAGNPS calculations the final water loading at the outlet was greater than the water yield generated from the direct surface runoff and the quick return flow. This implies that water is being added to the final outlet loading within the AnnAGNPS accounting features. Baseflow is not currently included in the model (Bingner et al., 2009), but the additional flow appears to closely mimic a groundwater supplement to the surface water flow. Similar to the average monthly streamflow from the Canard River subwatershed, Figure 5-10, February generally produced the highest monthly streamflow and July produced the least.

Table 5-10: Average Monthly Streamflow (mm)

Month	January	February	March	April	May	June	July	August	September	October	November	December
Monthly Streamflow	29	40	32	22	14	10	8	15	18	16	20	34

5.4.3 Sediment Results

The following section presents the AnnAGNPS Big Creek Watershed's model's sediment simulation results for the twenty year study period. The AnnAGNPS model is capable of tracking five different classes of particles (Table 3-1). Each soil type and layer is defined in terms of percent composition of each of the particle sizes. For the Big Creek Watershed, the soils, and thus also the sediment runoff, are composed of only three of the five particle classes: clay, silt and sand. The current sediment results presented have not been validated by field data.

In Figure 5-16 the average annual sediment yield and loading are outlined for the three soil particle types being tracked by AnnAGNPS in the Big Creek Watershed. All erosion presented in this section consists of only sheet and rill erosion, calculated by RULSE. Gully erosion was not included in the study. The average annual sediment yield is composed primarily of clay accounting for 52% of the total yield. The second and third largest components of the average annual sediment yield were silt and sand at 38 % and 10 %, respectively. The total average watershed sediment yield is 1.38 Mg/ha/yr. The modelled average annual sediment loading at the watershed's outlet was 0.612 Mg/ha/yr. The composition of the sediment loading was primarily clay at 85 %, silt at 14 %, and sand at approximately 1 %. The sediment loading at the watershed outlet is significantly less than the yield, a 55 % reduction. This reduction is intuitive as the modelled sediment would settle during the transport process, and that stream bed sediment erosion was not considered in the model.

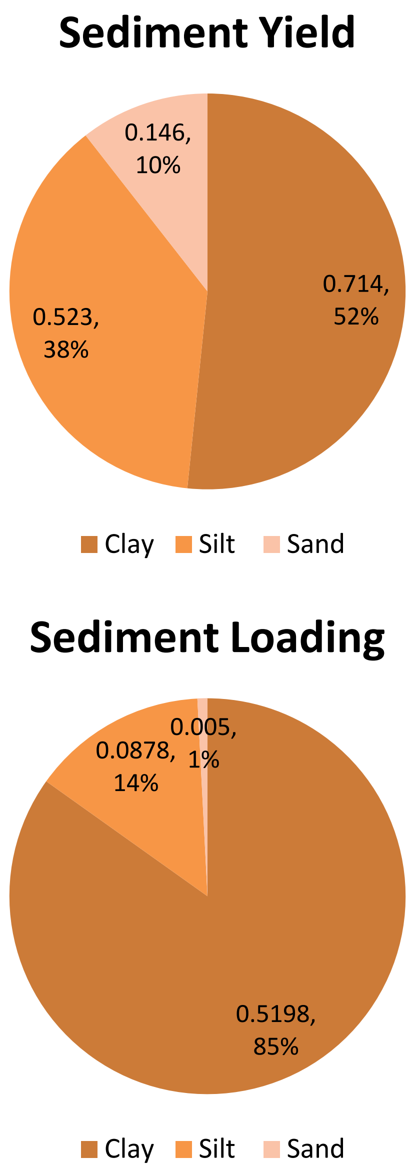

Figure 5-16: Average Annual Sediment Yield and Loading (Mg/ha/yr)

Figure 5-17 shows the spatial distribution of the sediment loading over the Big Creek cell network. Both the north-western and south-eastern regions of the Big Creek Watershed have higher sediment loading rates than the rest of the watershed. The higher rates are a function of the larger slopes in these regions compared to the center of the watershed. The areas of higher sediment loading generally also have a higher sediment erosion rate, indicating that the RUSLE K factor may have influenced the regions of higher sediment yield. The soil type composition also influences sediment loading at the outlet. The general areas producing more sediment are composed of Perth Clay and Brookston Clay. Clay particles have the lowest deposition ratios, and soils composed of high clay contents could produce high sediment loads.

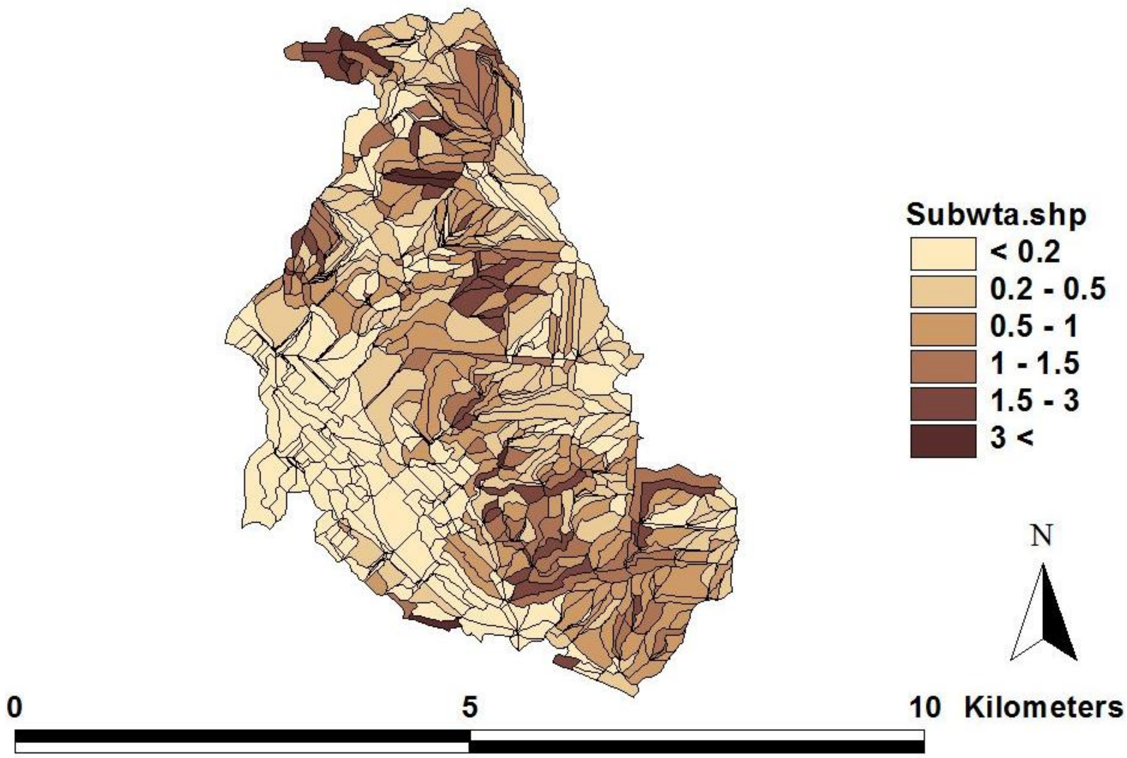

Figure 5-17: Spatial Distribution of Sediment Loading (Mg/ha/yr)

Summary statistics for the average annual sediment yield (Mg/ha/yr) from each of the 660 delineated cells are listed in Table 5-10. For clarification purposes, the sediment particle type distribution outlined in Figure 5-16 represents the average mass outflow per annum per unit area over the entire watershed. However, the statistics in Table 5-10 compare each cell's sediment yield distribution. Because the cells being analyzed in the same table all vary in size, the averages will not match the averages in Figure 5-16. The smallest sediment yields and loadings 0.00 Mg/ha/yr were from cells defined as open water. The highest sediment yield was generally produced in agriculture fields in the northern region of the watershed.

Table 5-11: Annual Average Sediment Loading and Yield (Mg/ha/yr)

	Loading				Yield			
	Clay	Silt	Sand	Total	Clay	Silt	Sand	Total
Average	0.538	0.090	0.001	0.629	0.745	0.562	0.166	1.474
SD	0.644	0.167	0.021	0.756	0.894	0.636	0.287	1.722
CV	1.196	1.862	13.774	1.200	1.200	1.131	1.727	1.169
Max	5.3	2.3172	0.5097	7.1765	7.284	6.604	5.397	16.305
Min	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

In Figure 5-18 the total daily sediment loading at the watershed outlet over the model twenty year simulation period is illustrated. The sediment loading in the same figure includes all three sediment classes modelled in the simulation. The maximum and minimum modelled daily flows were 2 230 Mg/day and 0 Mg/day, respectively. The modelled average daily mass flow of total sediment from the Big Creek Watershed is 10.8 Mg/day, and the data has a wide distribution with a coefficient of variation of 6.37. Table 5-12 contains the monthly average total sediment yield, with units of Mg/month. The months of April through June have the highest average monthly sediment yield rates, which is likely caused by spring runoff events.

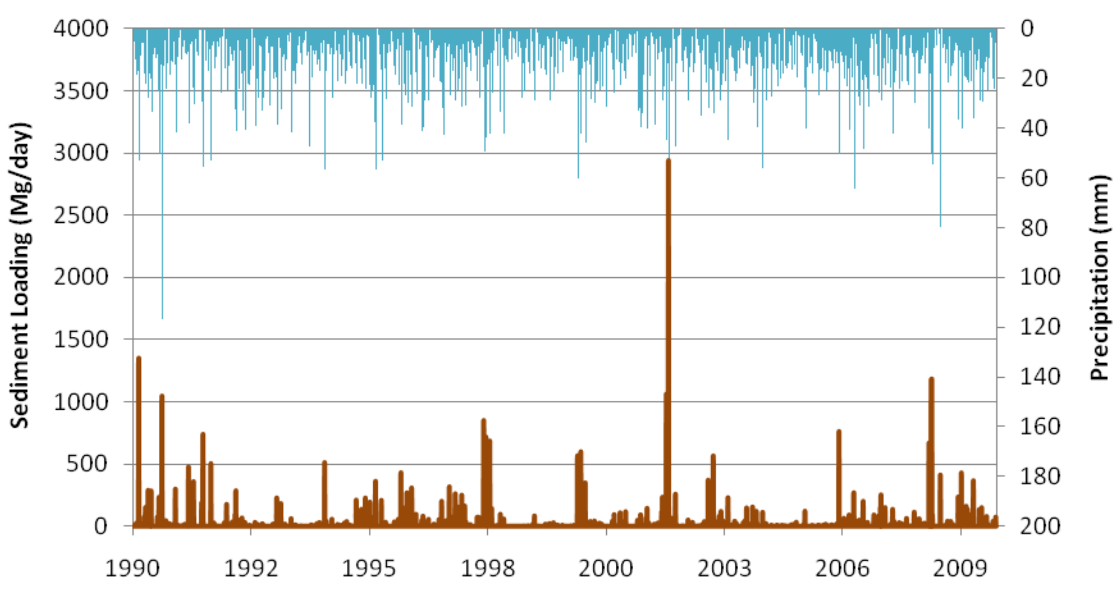

Figure 5-18: Total Sediment Loading

Table 5-12: Average Monthly Sediment Loading (Mg/month)

Month	January	February	March	April	May	June	July	August	September	October	November	December
Monthly Sediment Load	32	327	319	546	952	668	178	349	175	138	89	90

5.4.4 Nutrient Results

This section contains the nutrient results from the AnnAGNPS twenty year model simulation in the Big Creek Watershed. Though the model is capable of tracking organic carbon, phosphorus and nitrogen, only the latter two nutrients are considered in this investigation. Additionally, pesticides can be tracked in a similar manner to that of nutrients, but are also excluded. Both nitrogen and phosphorous exist in two forms: dissolved into a solution and attached to clay size particles. The procedures for tracking both dissolved and attached nutrients are based upon the concept of mass balances.

In Figure 5-19 and Figure 5-20 the annual average nitrogen yield and loading, and the annual average phosphorous yield and loading are outlined. The values in these figures represent the total average per area per annum from the watershed over the entire simulation period.

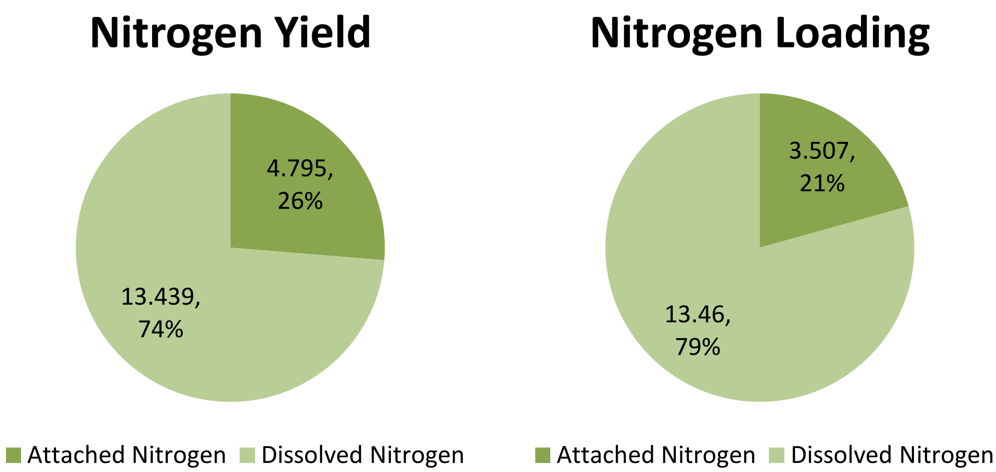

Figure 5-19: Average Annual Nitrogen Yield and Loading (kg/ha/yr)

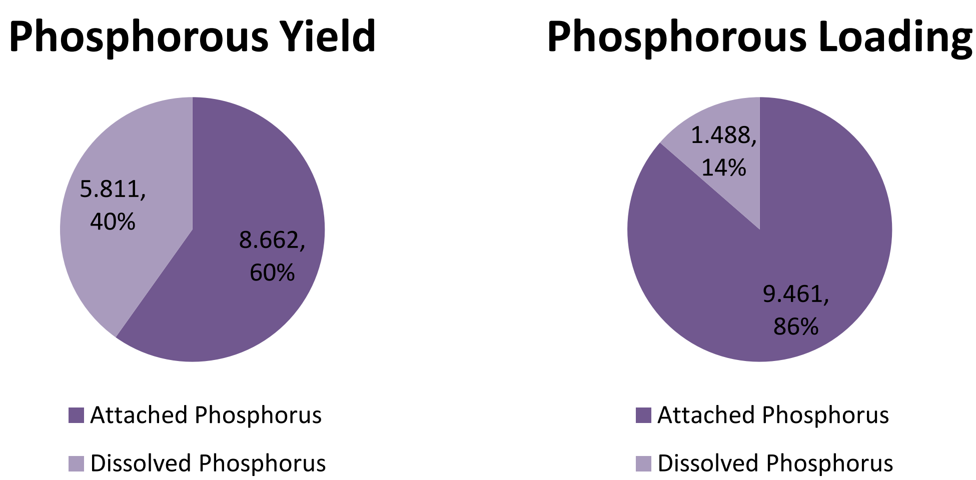

Figure 5-20: Average Annual Phosphorous Yield and Loading (kg/ha/yr)

The composition for both the nitrogen and the phosphorous annual average yield changed at the watershed outlet. Both nutrients in general deposited some mass during transport to the watershed outlet. For both nutrients, portions of the dissolved material is transformed in the reach or cell equilibrium process from dissolved to attached.

The following two figures, Figure 5-21 and Figure 5-22, show the spatial distribution of the Big Creek Watershed's average annual total nitrogen and phosphorus loading per cell. The average annual total yield represents the mean yearly kilograms of nutrients per hectare (both dissolved and attached) that originate from a cell and leave the watershed outlet.

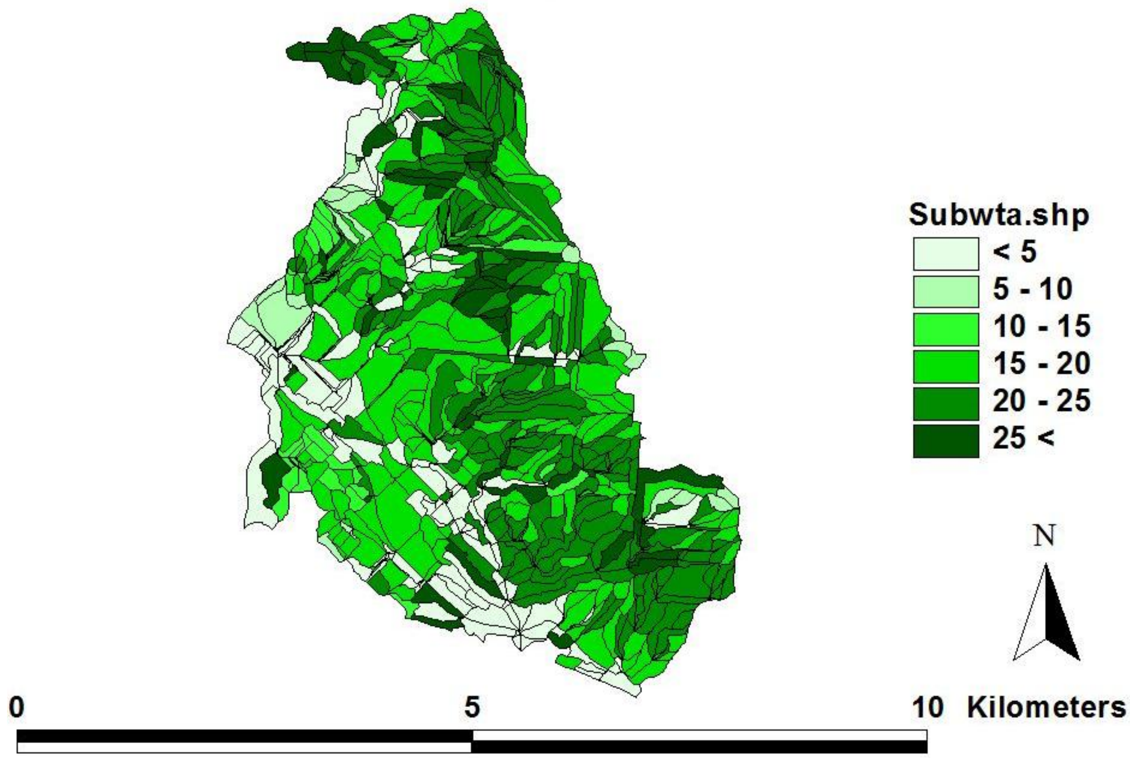

Figure 5-21: Spatial Distribution of Nitrogen Loading (kg/ha/yr)

The average annual yield for nitrogen for the whole watershed is approximately 16.8 kg/ha/yr, with a minimum cell value of 0 kg/ha/yr and maximum cell value of 54 kg/ha/yr. The cells with the lowest average annual nitrogen yields in the watershed have landuse types of forest or open water. The highest nitrogen average annual yields correspond to the agricultural landuse cells. Additionally, the average annual phosphorus yield for the entire watershed is 11.4 kg/ha/yr, with a minimum cell value of 0 kg/ha/yr and maximum cell value of 71 kg/ha/yr. Similar to the nitrogen yields, the cells with the lowest average annual phosphorus yields in the watershed had landuse types of forest and open water.

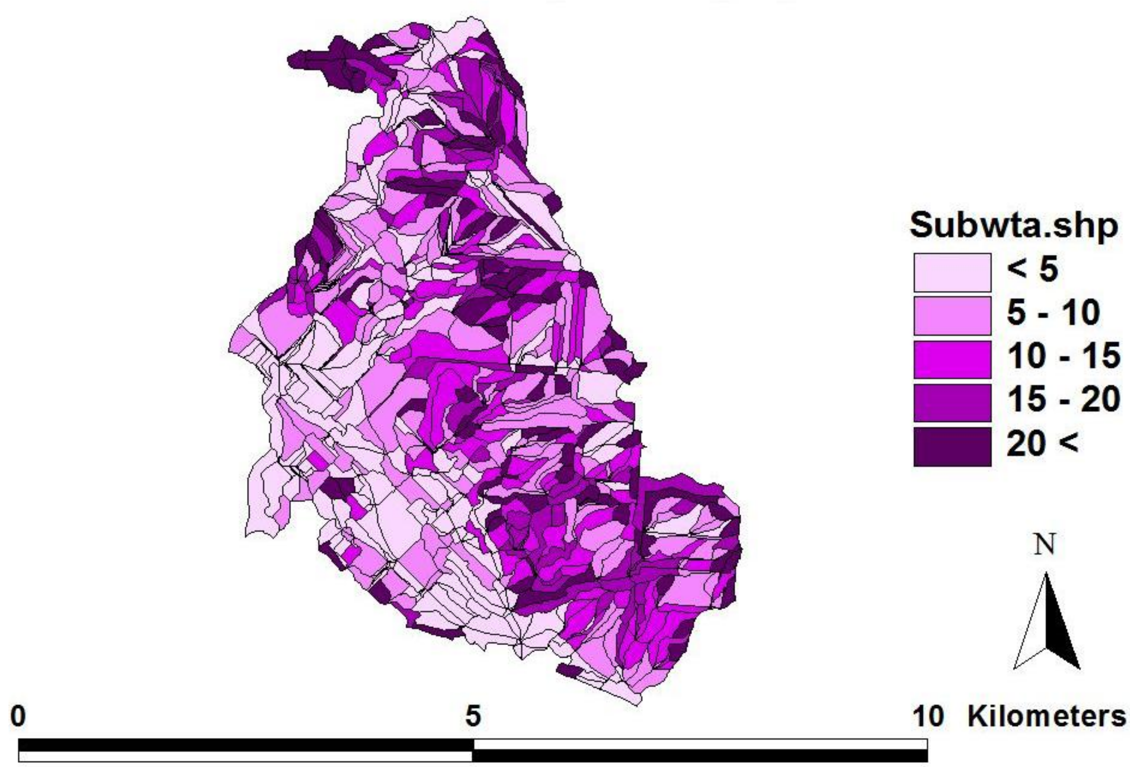

Figure 5-22: Spatial Distribution of Phosphorous Loading (kg/ha/yr)

Summary statistics for the average annual nutrient yield from each of the 660 delineated cells from the AnnAGNPS Big Creek Watershed model are listed in Table 5-13. The same table also outlines the average, standard deviation, coefficient of variation, maximum value and minimum value for the attached, dissolved and total nutrient yield for both nitrogen and phosphorus. The units in the table are kg/ha/yr. As previously noted in the preceding section, the statistics outlined represent the variation between individual cell's average results. Due to the variation between the individual cell sizes, the averages in Table 5-13 are not the same as the average total for the whole watershed.

Table 5-13: Annual Average Nutrient Loading (kg/ha/yr)

	Attached Phosphorus	Dissolved Phosphorus	Average Phosphorus	Attached Nitrogen	Dissolved Nitrogen	Total Nitrogen
Average	10.1	1.4	11.4	3.7	13.1	16.8
SD	4.2	6.5	8.8	4.2	6.5	8.8
CV	0.4	4.8	0.8	1.2	0.5	0.5
Max	68.3	6.6	71.3	36.8	26.8	54.2
Min	0.0	0.0	0.0	0.0	0.0	0.0

Figure 5-23 and Figure 5-24 illustrate the total nitrogen and phosphorus mass daily flow rates over the modelled twenty year simulation period. The total nutrient yield, phosphorus or nitrogen, includes both the attached and dissolved nutrient components. The maximum and minimum modelled daily total nitrogen flows are approximately 19.5 Mg/day and 0 Mg/day, respectively. Similarly, the maximum and minimum modelled daily total phosphorus flows are approximately 31 Mg/day and 0 Mg/day, respectively. The average daily mass flow of nitrogen and phosphorus from the Big Creek Watershed are 216 kg/day and 135 kg/day, respectively.

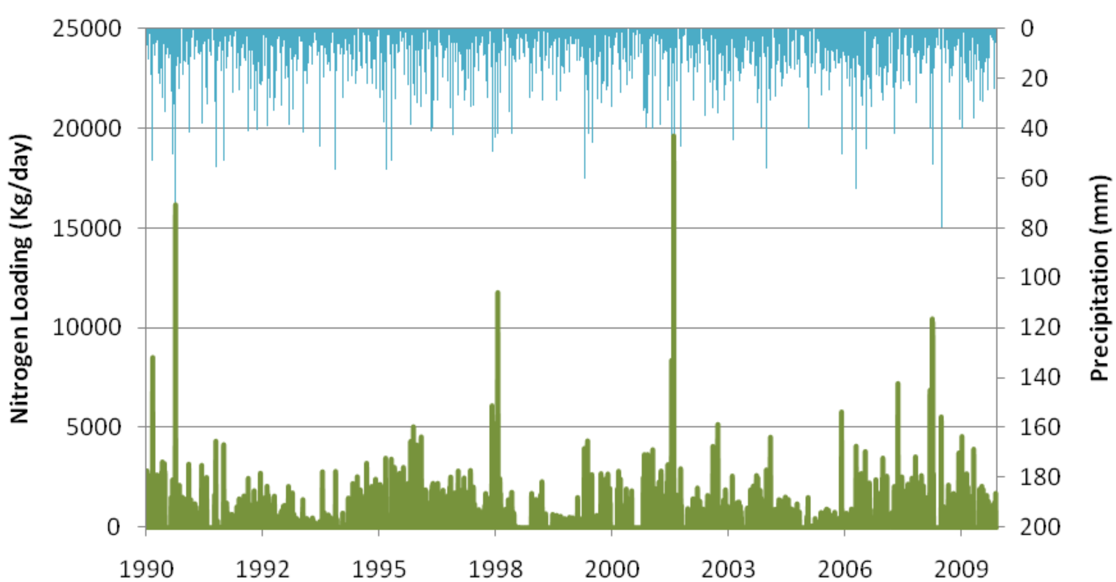

Figure 5-23: Total Nitrogen Loading

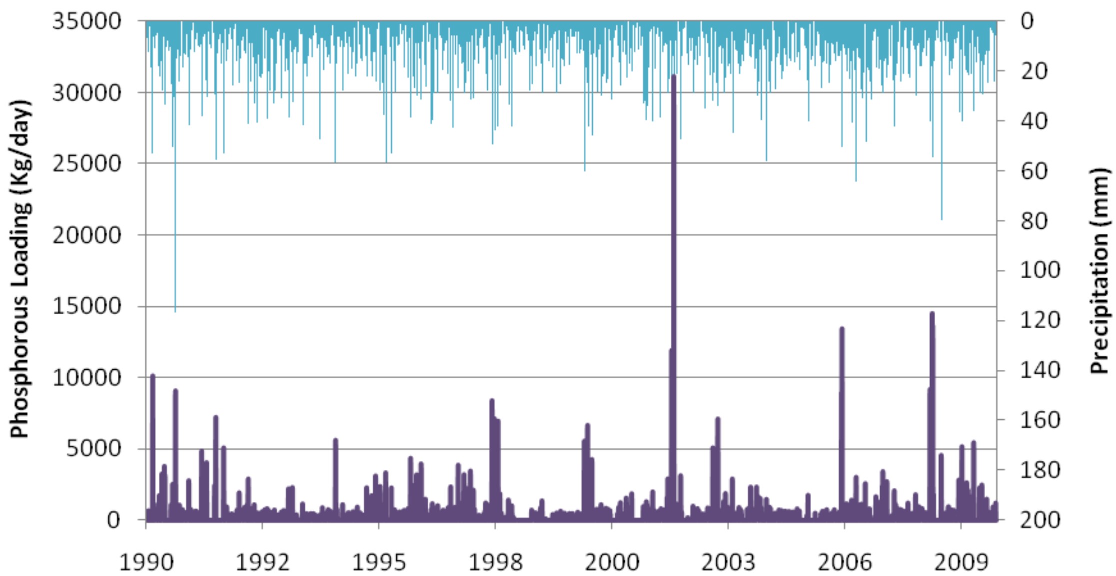

Figure 5-24: Total Phosphorous Loading

Table 5-14 provides the monthly average total nutrient yield. The units for this table are kg/month. The months of March through May had the highest average monthly sediment yield rates, which is likely caused by the spring runoff events, and by fertilizers added to new crops during this time.

Table 5-14: Average Monthly Nitrogen Phosphorous Loading (Mg/month)

Month	January	February	March	April	May	June	July	August	September	October	November	December
Total Nitrogen Load	7.1	9.7	11.5	14.7	12.4	8.1	4.6	7.2	8.7	9.0	7.6	8.5
Total Phosphorus Load	2.6	5.7	7.0	9.5	13.7	8.8	3.2	5.1	3.7	3.8	3.4	3.8

5.5 Summary

This chapter provided a summary of the AnnAGNPS database used in the watershed modelling. The summary included a review of the meteorological datasets, a brief outline of GIS data utilized in the model, and a description of the text database utilized in the modelling. Observed data for calibration and validation was not readily available for the Big Creek Watershed, the region the study was focused on. However, a neighbouring watershed, Canard River, had over 35 years of recorded streamflow data publicly available. The AnnAGNPS model's database was calibrated and validated in the Canard River Watershed with two eight year periods of daily streamflow data. The monthly efficiencies in the Canard River Watershed for both periods were generally good to very good.

Eight months of observed streamflow in the Big Creek Watershed were utilized to verify the model's efficiency within the study watershed. Comparing the limited streamflow data and the AnnAGNPS model predictions the Nash-Sutcliff efficiency and the coefficient of determination ranged from good to poor in the monthly and daily reviews. To further verify the model within the Big Creek Watershed addition streamflow data could be collected. The limited time span of the recorded data provided only a limited scope for verification.

The average monthly trends in the Big Creek Watershed corresponded closely with the monthly trends in the Canard River Watershed. An extensive summary of the Big Creek AnnAGNPS model outputs were outlined in the later section o this chapter. In general the regions producing a higher sediment and nutrient yield were in the north-eastern and southern regions of the watershed. To continue the investigation of the model's adequacy in the Essex Region, the AnnAGNPS model could be calibrated and validated using observed continuous sediment or nutrient data. This would allow for verification of alternative model outputs which have not currently been investigate studied in the Ontario Region.

CHAPTER 6: BIG CREEK MARSH WATER BUDGET MODEL RESULTS

6.1 Introduction

The following chapter outlines the Big Creek WB model results over the forty year study period (1969-2008). A review of the three Marsh operating scenarios will be evaluated with the WB model. The three years of recorded pumping data will be evaluated under each operating scenario. A single variable sensitivity analysis will be undertaken with several of the model input variables. The chapter will also include an investigation of the effect of Lake Erie's water levels on the Marsh's hydrological state.

6.2 Big Creek Marsh Operating Scenarios

In the Big Creek Marsh WB model development the governing mass balance equation (Equation 4.1) was outlined. In the same equation only selected flows in the WB are independent of other model variables, with the independent variables being streamflow and precipitation. All other WB components are a function of at least one other variable in the model. Three components of the wetland WB are dependent on the operating scenario: pumping into the Marsh, pumping of the Marsh, and flow released from the control dam. Thus, to properly evaluate the wetland WB the appropriate operating phase or phases must be implemented to accurately estimate the three aforementioned WB flows.

There are three distinct vegetative and hydrologic phases proposed in the permit to take water (PTTW) (Ducks Unlimited Canada, 2007): open water, hemi, and overgrown (Table 4-1). The open water phase has the lowest water levels of the three, requiring pumping out of Big Creek as a standard operation, not a contingency. The intent of this phase is to promote the growth of new flora that thrives under drier conditions. The overgrown phase is the opposite of open water, where the wetland is flooded with higher water levels. The intent of this phase is to inundate the wetland to terminate some of the existing vegetation. The hemi phase is an intermediate state between the open water and overgrown phases. It is also the most productive phase for the majority of wetlands. This ecological state is most likely able to support the highest diversity of species. Consequently, the hemi phase under ideal operating scenarios is maintained for a much longer duration than the open water or overgrown phase.

Due to the lack of operational data, the years with recorded data, 2006 to 2008, will be modelled under all three scenarios to investigate which operating scenario was likely implemented each year. Figures 6-1, 6-2, and 6-3 illustrate the major hydrologic components of all three years under each operating phase. In the same figures two pumping alternatives are compared. The first alternative outlines the model results with the pumping algorithm implemented in the simulation. The second alternative replaces the pumping algorithm in the model with the recorded pumping data.

In the alternative with the recorded pumping data all other model parameters remain the same as in the first alternative with the exception of disabling the gate release outflow during and between short breaks in the recorded pumping. In the hemi and overgrown phases, pumping out of the Marsh to Lake Erie is not included in the budget calculations, as it was only outlined as a contingency operation (Ducks Unlimited Canada, 2007). Pumping out of the Marsh is part of the normal operations under the open water scenario and is not regarded as contingency. Therefore the pumping out of the Marsh was included in the WB calculations for the open water phase. The timing and the magnitude of outflow pumping is included in the open portions of Figures 6-1to 6-3.

The same figures had the variables of lesser significance removed to make the figures more legible. Since streamflow into the Marsh from the watershed and precipitation are directly related, only streamflow is depicted. Seepage inflow and outflow components are also not included since they account for 1% or less of total inflow or outflow respectively in all three operating phases.

6.2.1 Marsh Operations 2006

This section of the chapter will review the Big Creek Marsh operations in 2006. In Figure 6-1 the open water, hemi, and overgrown phase annual WBs are compared under two alternatives, the modelled pumping and the recorded pumping.

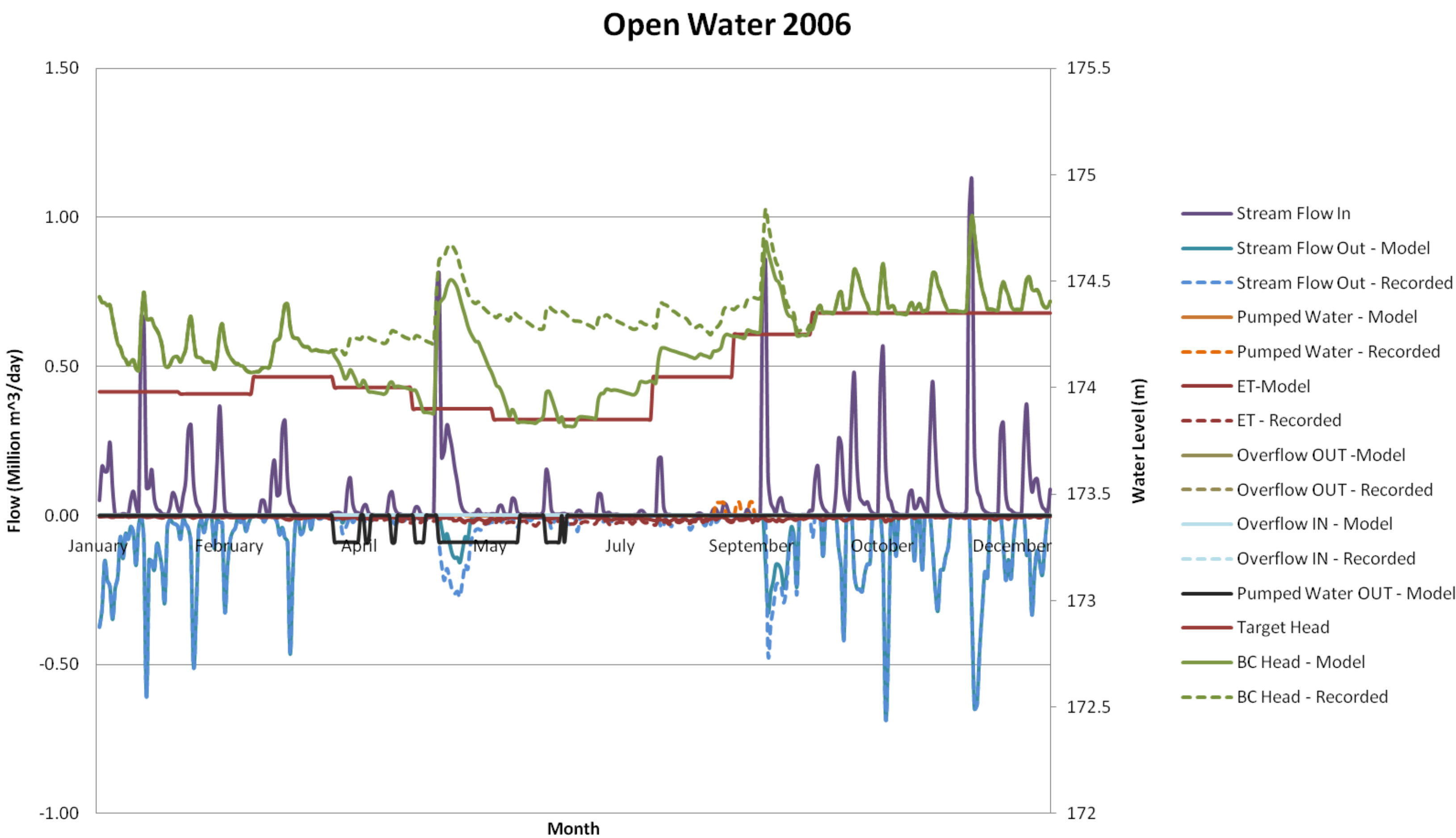

Figure 6-1: 2006 Hydrological Flows (a) Open Water Marsh Operations

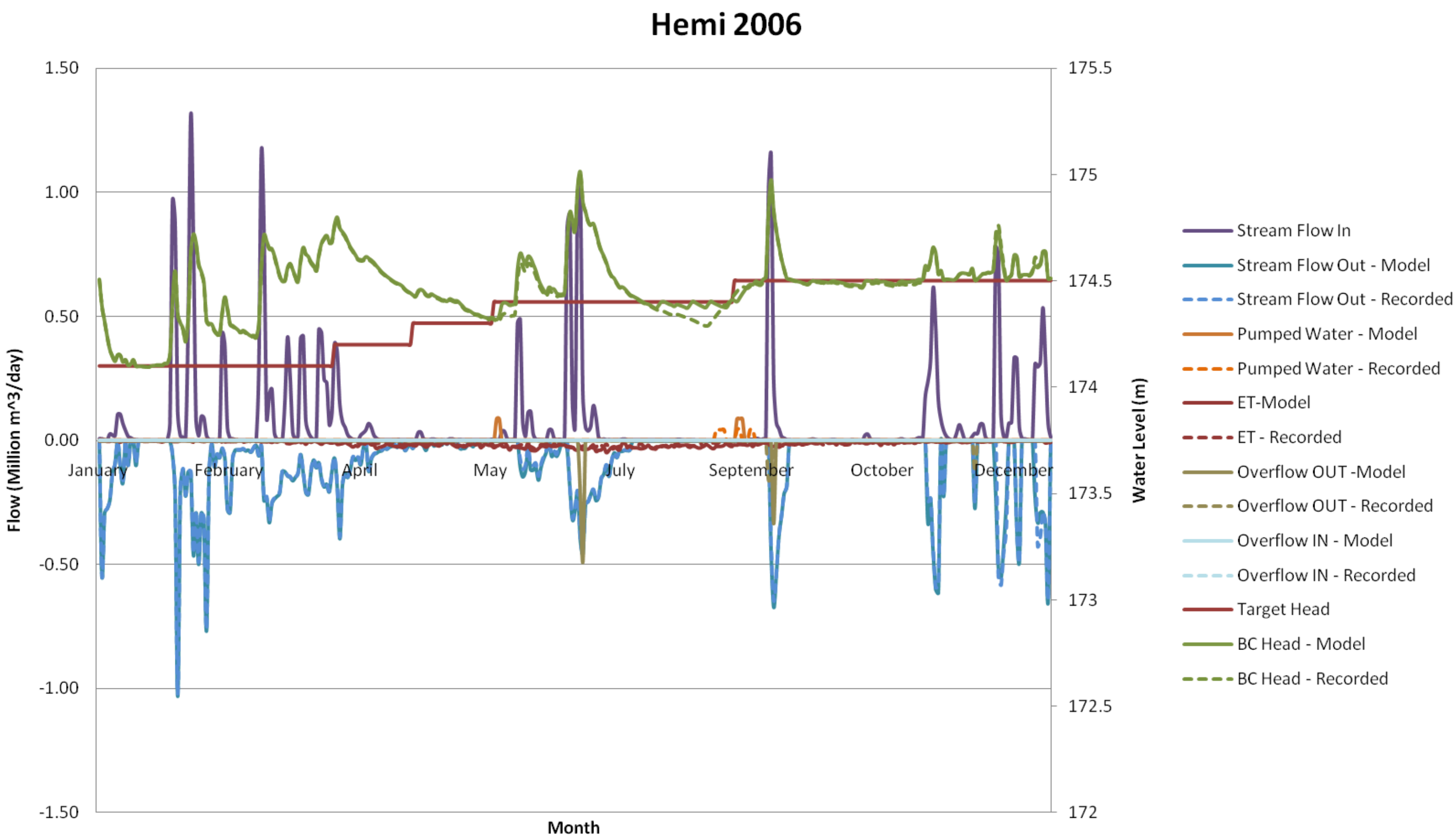

Figure 6-1: 2006 Hydrological Flows (b) Hemi Marsh Operations

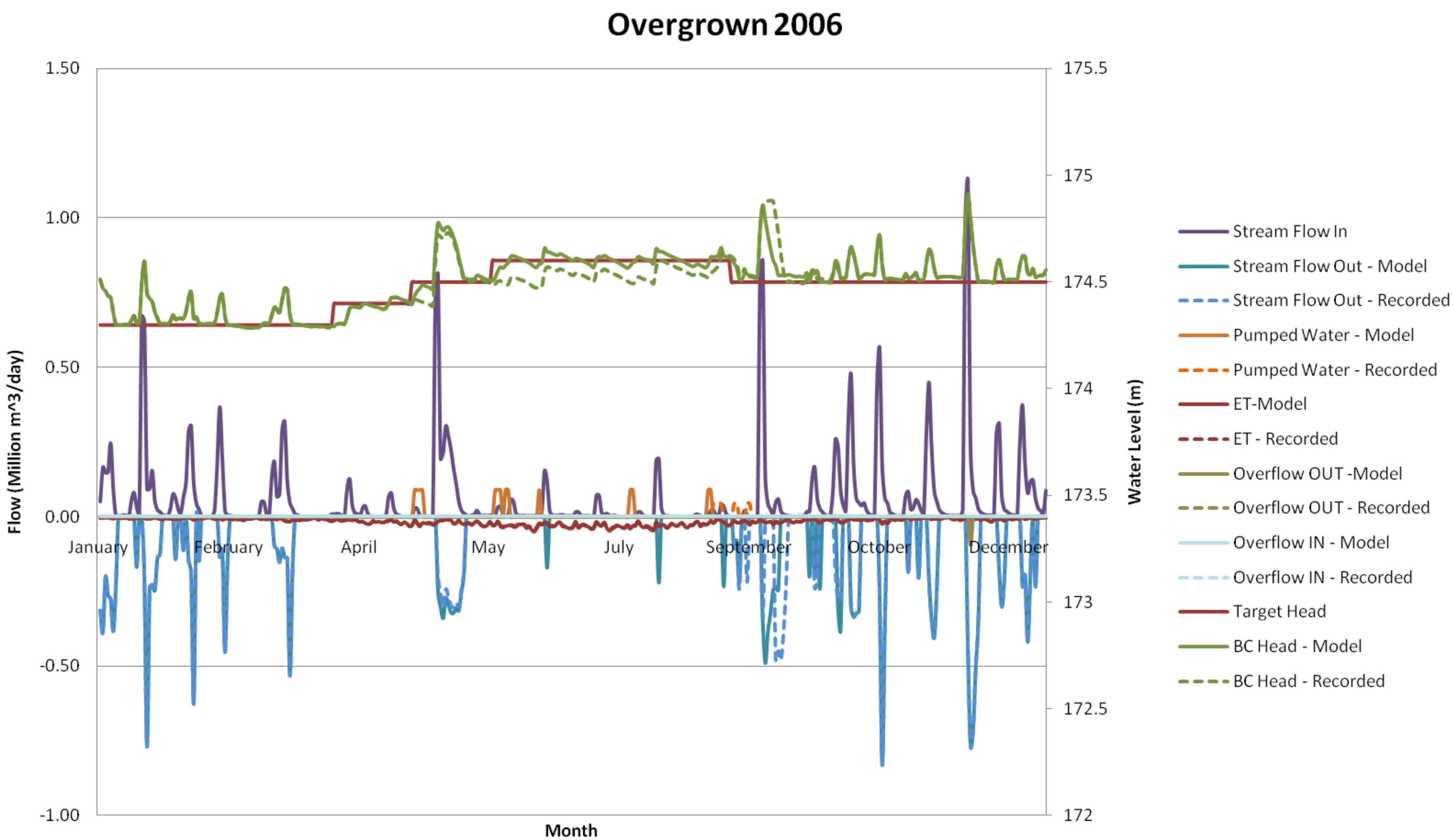

Figure 6-1: 2006 Hydrological Flows (c) Overgrown Marsh Operations

In the WB analysis of all three operating phases for 2006 the water level targets can generally be met in the model pumping alternative as shown in Figure 6-1. However, not all three phase targets can be met in the recorded pumping alternative. The recorded pumping data indicated inflows in September but according to the PTTW (Ducks Unlimited Canada, 2007) the prescribed pumping season for the overgrown phase does not include the month of September. This would suggest the overgrown phase was not implemented in 2006.

In the WB analysis of the open water phase for 2006 the recorded pumping water levels would not be near the goals of the operating plan with recorded pumping inflow to the Marsh. In order to meet the operating water level goals of the open water plan pumping out of the Marsh is necessary. Since the recorded pumping appears to contradict the operation plan of the open water phase, it would be reasonable to assume this phase was not implemented in 2006.

In the WB analysis of the two alternatives from a visual inspection of the hemi operating phase the modelled water level and the recorded water level generally match closely. Both alternative water levels spike in June because of the high stream inflow. The recorded pumping alternative's water levels fall below the operating phase's target levels in late August. In a comparison of all three operating plans the two alternatives in the hemi phase operations have the least difference between the two water levels. This would suggest the hemi phase is the most likely operating plan in 2006.

6.2.2 Marsh Operations 2007

The following section reviews the Big Creek Marsh operations in 2007. In Figure 6-2 the open water, hemi, and overgrown phase annual WBs are compared under two alternatives, the modelled pumping and the recorded pumping.

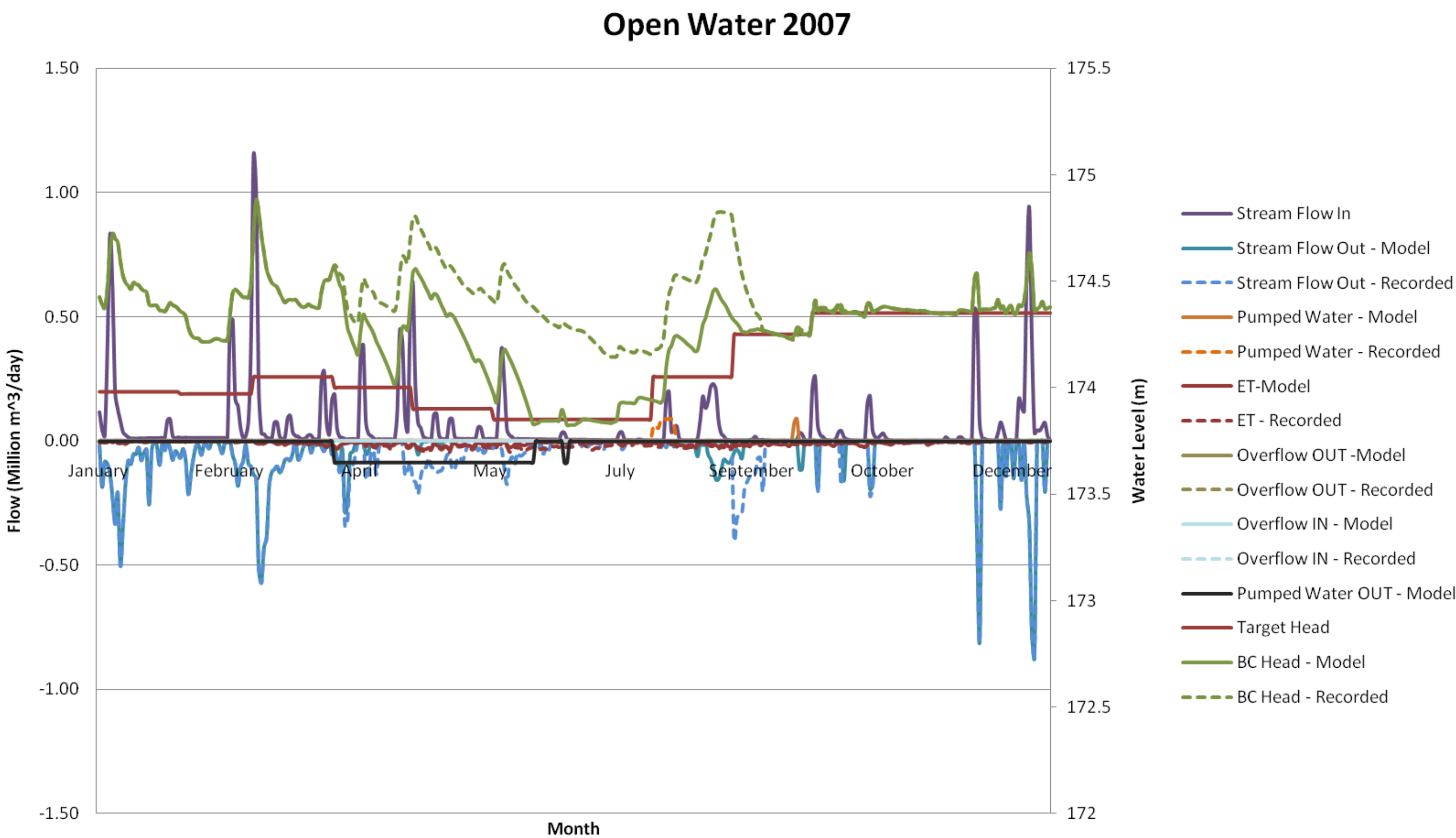

Figure 6-2: 2007 Hydrological Flows (a) Open Water Marsh Operations

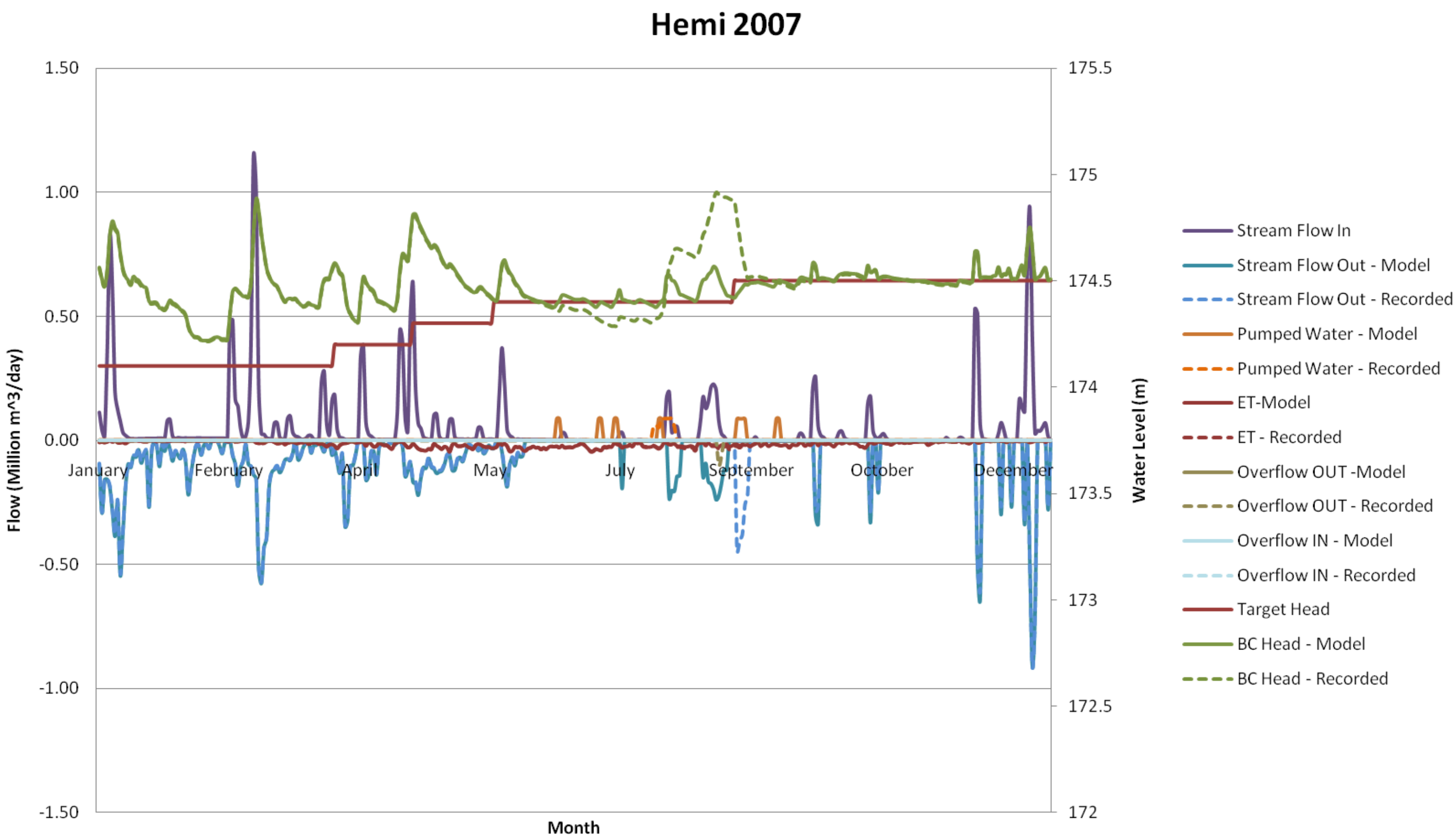

Figure 6-2: 2007 Hydrological Flows (b) Hemi Marsh Operations

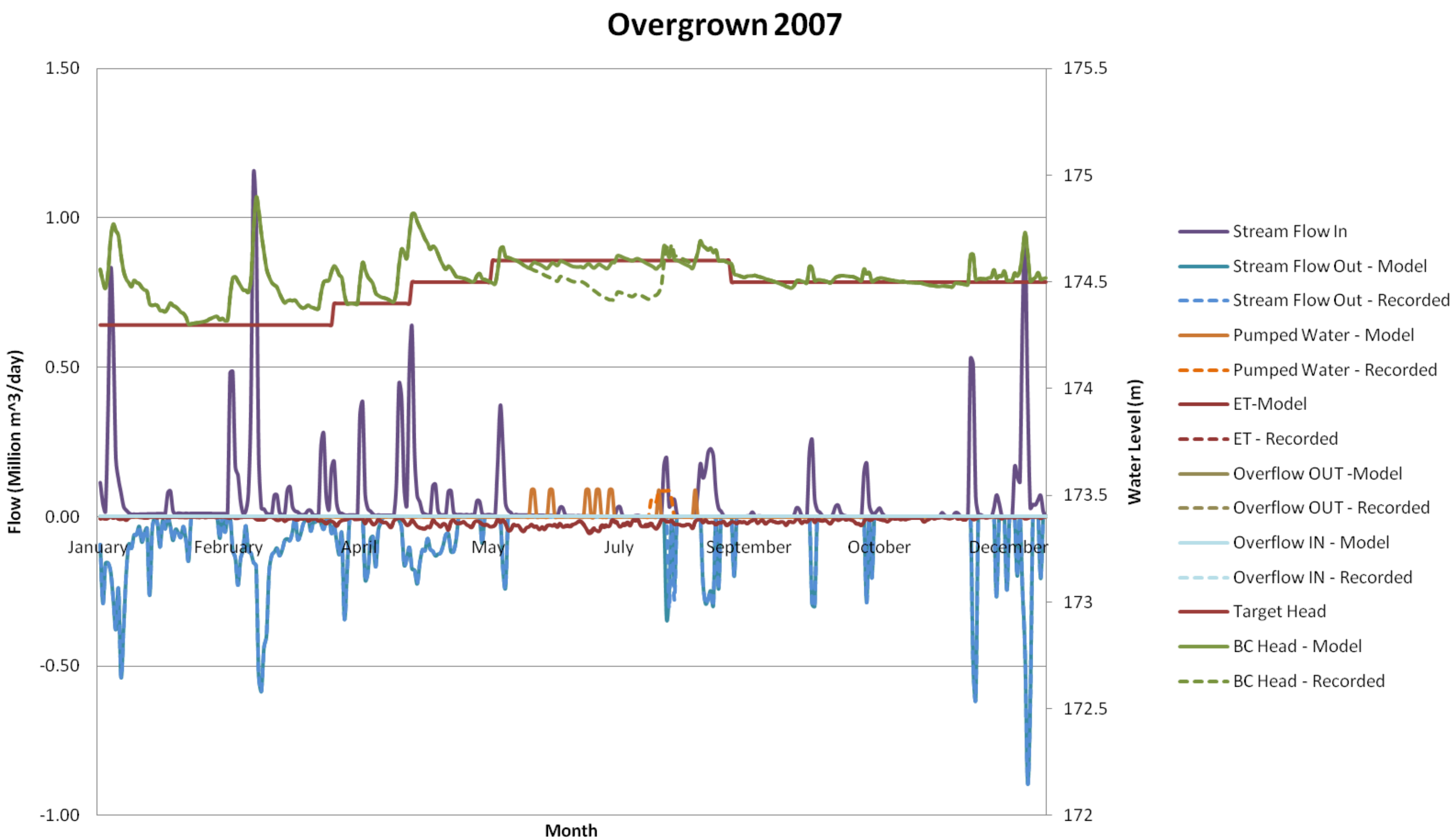

Figure 6-2: 2007 Hydrological Flows (c) Overgrown Marsh Operations

In the WB analysis of all three operating phases for 2007 the water level target goals can generally be met in the model pumping alternative as outlined in Figure 6-2. The open water phase's water level targets are not met when the recorded pumping is implemented in the WB model. This would suggest the open water operating phase was not employed in 2007. Both the overgrown and hemi phase modelled and recorded pumping alternatives reasonably match the targets. The overgrown modelled and recorded pumping water levels closely overlap.

In August 2007 recorded pumping into the Marsh occurs at the same time as a sizable rainfall and streamflow event. This appears out of place as it is expected that pumping would be minimized or halted during and shortly after a storm event until all the naturally draining water has entered into the wetland. This may imply that the overgrown operating phase was implemented in 2007. The hemi phase recorded pumping alternative does not match the operating target water levels well in the month of August.

In the 2007 hemi simulation, the timing of the modelled and recorded pumping is inconsistent. The 2007 model pumping occurs periodically with a fairly even distribution throughout the prescribed pumping season, whereas the recorded pumping data occurs as a single large influx in August. The recorded pumping in 2007 occurs simultaneously as a large streamflow event enters the Marsh.

6.2.3 Marsh Operations 2008

This section of the chapter will review the Big Creek Marsh operations in 2008. In Figure 6-3 the open water, hemi, and overgrown phase annual WBs are compared under two alternatives, the modelled pumping and the recorded pumping.

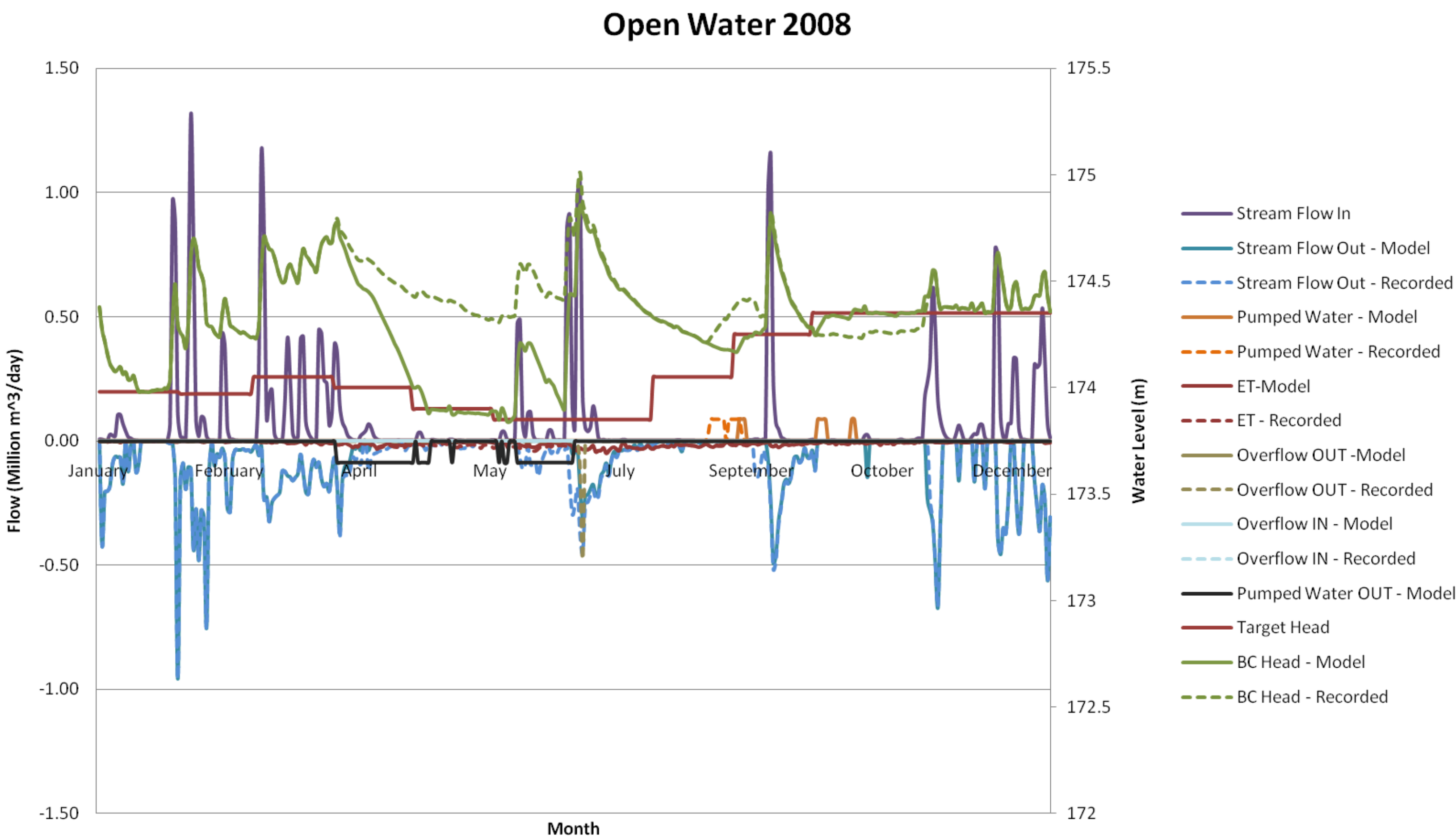

Figure 6-3: 2008 Hydrological Flows (a) Open Water Marsh Operations

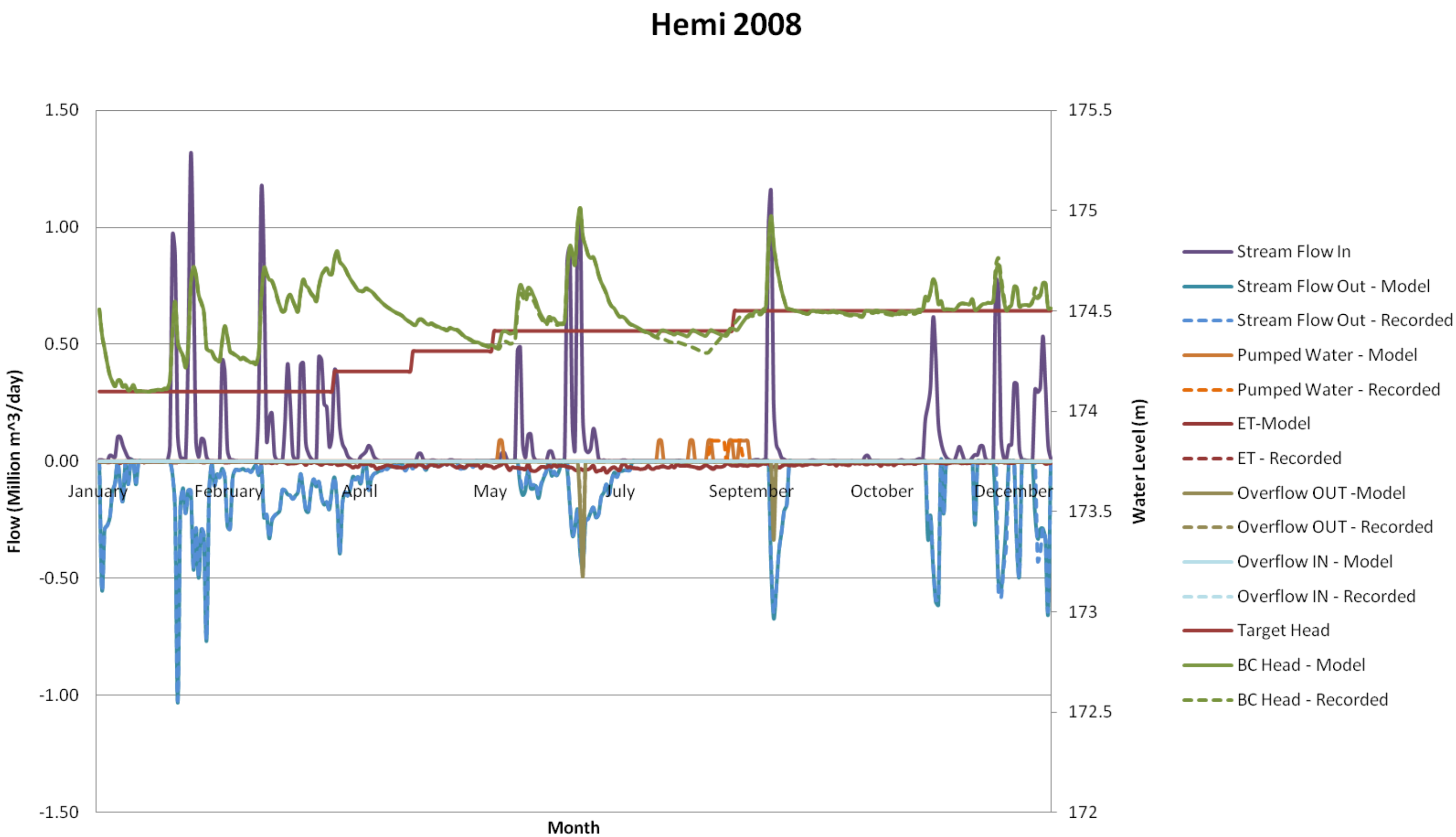

Figure 6-3: 2008 Hydrological Flows (b) Hemi Marsh Operations

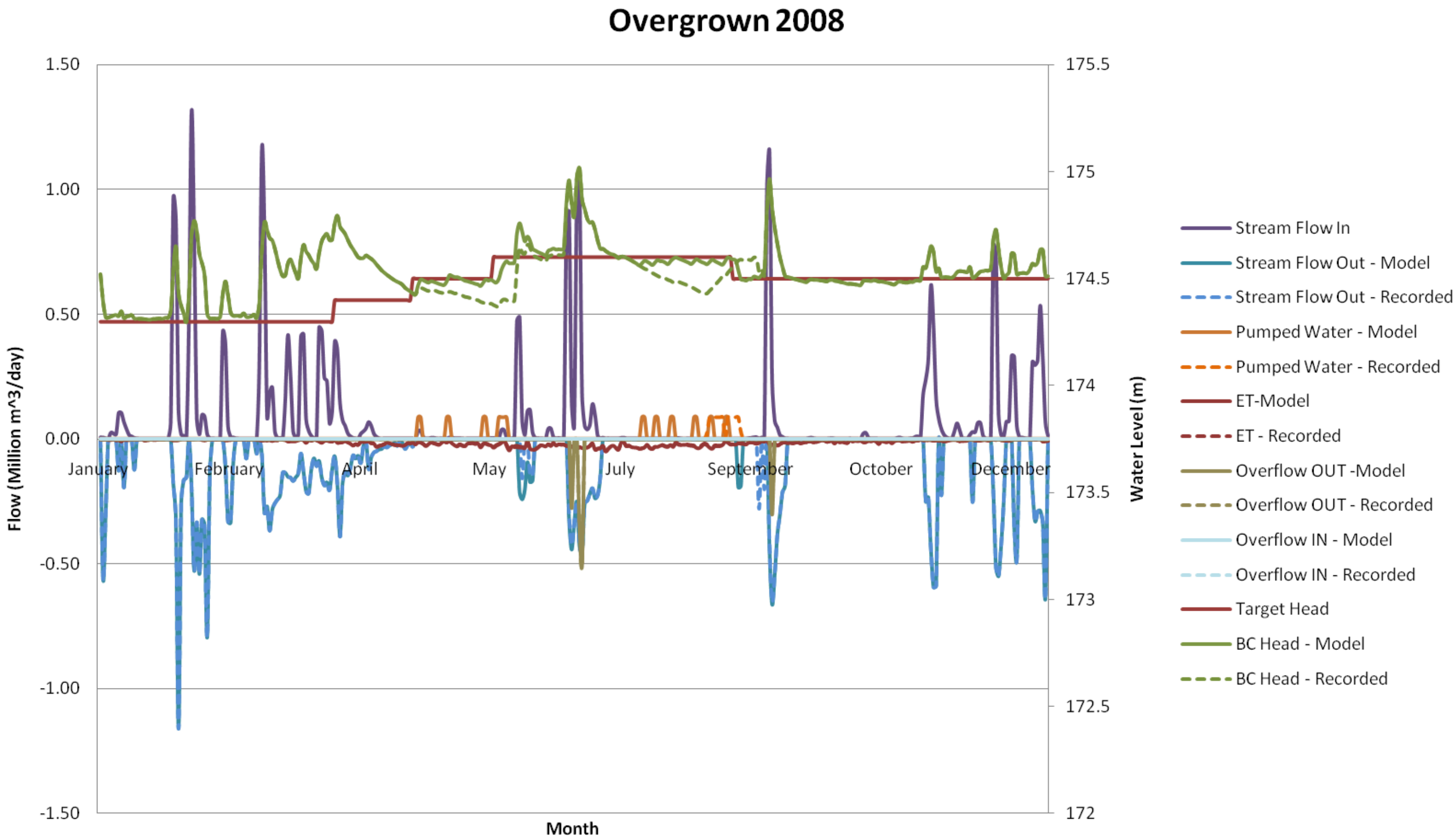

Figure 6-3: 2008 Hydrological Flows (c) Overgrown Marsh Operations

In the 2008 WB analysis the hemi and the overgrown operating phases' target water levels can generally be met in the modelled pumping alternative as outlined in Figure 6-3. Similar to 2006 and 2007 the open water phase's water level targets are not met in 2008 when the recorded pumping is implemented in the WB model. This would suggest the open water operating phase was not executed in 2008. In the 2008 recorded pumping data inflow pumping to the Marsh is included in September. In accordance with the PTTW (Ducks Unlimited Canada, 2007) pumping in the month of September is not prescribed under normal operations. This would suggest that the operating plan in 2008 was not the overgrown phase.

In the hemi WB alternatives both the modelled pumping and the recorded pumping water levels match each other fairly well, suggesting that the hemi phase could have been operated in 2008. Coupled with that the fact both the overgrown and open water phases do not appear to reasonable fit with the recorded pumping, this strongly suggests the hemi phase operation plan was implemented in 2008.

6.2.4 Marsh Operations Scenarios Summary

This section will summarize and review each one of the three proposed phase's hydrologic goals, consider the phases' feasibility given the actual pumping and outline the potential of each phase in each year with observed data.

In Table 6-1 the monthly total pumping inflow into the Marsh is compared for all three operating scenarios for the years that have recorded data (2006-2008). In addition the recorded monthly pumping data is also summarized in Table 6-1. The data is lumped into monthly flows to simplify the table. The recorded pumping only represents inflow into Big Creek Marsh. Data for pumping out of the Marsh was not available. Without data for outflow pumping it cannot be concluded if outflow pumping did occur in these three years. If outflow pumping did not occur this would likely infer that the operating scenario for these years was not the open water phase.

Table 6-1: Comparison of Pumping Data (m^3)

Year	Month	Recorded Data	Open Water	Hemi	Overgrown
2006	May	-	-	-	353,480
	June	-	-	176,740	530,220
	July	-	-	0	176,740
	August	174,414	-	0	176,740
	September	174,413	0	265,110	-
	October	-	0	-	-
	Annual Sum	**348,826**	**0**	**441,850**	**1,237,180**
	Recorded Data % Difference	**-**	**-100**	**27**	**255**
2007	May	-	-	-	0
	June	-	-	176,740	353,480
	July	-	-	353,480	530,220
	August	658,336	-	176,740	176,740
	September	-	161,754	530,220	-
	October	-	0	-	-
	Annual Sum	**658,336**	**161,754**	**1,237,180**	**1,060,440**
	Recorded Data % Difference	**-**	**-75**	**88**	**61**
2008	May	-	-	-	530,220
	June	-	-	176,740	353,480
	July	-	-	0	176,740
	August	810,293	-	706,960	883,700
	September	203,482	265,110	353,480	-
	October	-	530,220	-	-
	Annual Sum	**1,013,774**	**795,330**	**1,237,180**	**1,944,140**
	Recorded Data % Difference	**-**	**-22**	**22**	**92**

In 2006 and 2008, the hemi phase has the smallest percent difference with the recorded data compared to the other phases, but has the highest difference in 2007. In terms of overflow for the hemi operating phase, a small overflow out of the Marsh is predicted once at the end of fall in 2006 and twice during two separate storm events in

2008. The open water phase had the same magnitude of percent difference for pumping inflow as the hemi phase in 2008, but the open water phase had pumping in the month of October, which was not recorded in the actual pumping data.

In both 2006 and 2008, the overgrown phase's modelled pumping has the highest annual average percent difference with the recorded data. These inconsistencies strongly suggest that the overgrown phase was not in operation in 2006 or 2008. However, the overgrown phase has the smallest percent difference in 2007, and there is not any recorded pumping occurring in the month of September. It appears likely that the Marsh was operated under an overgrown phase scenario in 2007. To maintain the high summer water levels suggested by the overgrown phase plan, significant pumping is required. From 2006 to 2008, the modelled pumping for the overgrown operations requires 60% to 250% more pumping than the recorded data. Similar to the hemi phase, overflow out of Big Creek overflow occurs in both 2006 and 2008.

The hemi phase corresponds well with the limited pumping data. The timing of the recorded pumping tends to occur when the hemi operation plan suggests increasing the Marsh's water level at the end of the summer. In 2006 and 2008 the model and recorded pump data closely overlap, both in terms of the timing and the magnitude, for the early fall period (Table 6-1). However, in the modelled hemi pumping scenario for all three years, pumping at the beginning of summer is required to increase the water level in accordance with the plan, which is not reflected in the recorded data. For the hemi phase, the modelled annual average pumped inflow is greater than the recorded pumping data for all three years. This suggests that the water levels prescribed in the operations plan may not have been fully realized.

A detailed review of the Big Creek WB results was undertaken (Section, 6.3), however, only one of the three operating phases, the hemi phase, was considered for the review. Exclusive of the above discussed pumping data, information related to actual historic wetland operating procedures is not currently available. Therefore, the methodology employed by the Big Creek Marsh operators had to be assumed. The hemi phase is the most logical choice since it is an intermediate phase suitable for average years that would likely be maintained more frequently. In addition this operating phase matches the best with two of the three years of recorded pumping data.

6.3 Hemi Water Budget

The present section contains a detailed review of the Big Creek Marsh hemi WB analysis over a forty year period from 1969 to 2008. This section of the chapter includes a general review of each WB flow and the Marsh storage. Outlined in the later portions of this section are a sensitivity analysis and an investigation of Lake Erie's influence on the Marsh's hydrological status.

With the hemi phase being considered the most appropriate of the three potential phases, the following section encompasses an in-depth review of the hemi WB. Figures 6-4 and 6-5 represent the average annual WB inflows and outflows in thousands of m^3 per year. The average annual inflows are nearly equal to average annual outflows. The resulting difference is the net change in storage over the forty year study period. The most significant inflow was streamflow, representing over 72 % of total inflow into the Marsh, with precipitation, overflow into the Marsh, pump inflow and seepage inflow at 21%, 5%, 2% and less than 1%, respectively. The most significant outflows were gate released outflow at 66 %, with evapotranspiration (ET), overflow and seepage outflow at 19%, 15% and less than 1%, respectively.

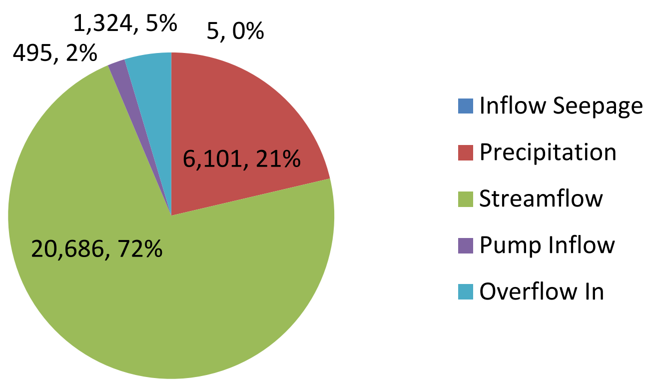

Figure 6-4: Water Budget Average Annual Inflows (thousand of m^3/year, %)

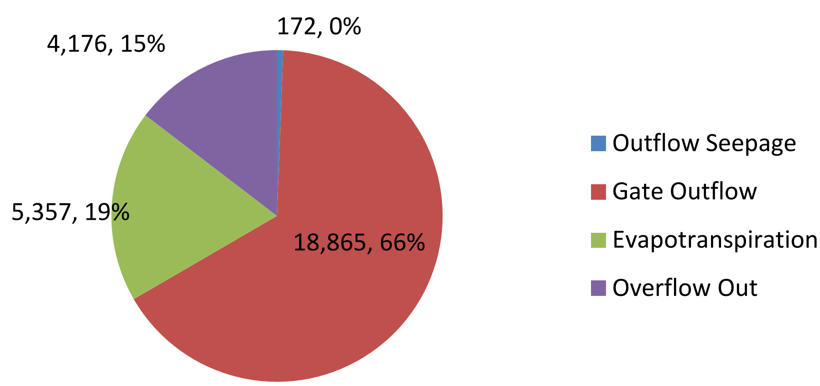

Figure 6-5: Water Budget Average Annual Outflows (thousand of m^3/year, %)

Summary statistics for the annual average WB components are listed in Table 6-2. The units in the table are m^3/year for the budget parameters and m above a datum of 173.7 m above mean sea level (AMSL) for the Big Creek Marsh water level. The Marsh's water level is included in the summary table to provide some context to average yearly storage quantities. The statistics presented in the table include the standard deviation (SD), coefficient of variation (CV), maximum value (Max), and minimum value (Min) of each WB components annual sum. The standard deviation is a measure of the variation each set has from its mean. A higher standard deviation indicates a higher variation in the data. The coefficient of variation is a normalized standard deviation that has been divided by the subject data set's mean. With this normalized statistic comparisons of the variation between data sets can be more equitably contrasted.

Table 6-2: Summary of Annual WB Statistics

	Big Creek Water Level (m above 173.7 m AMSL)	**Precipitation** (m^3/year)	**Streamflow** (m^3/year)	**Gate Outflow** (m^3/year)	**Seepage In** (m^3/year)
Average	**0.862**	**6,100,712**	**20,685,841**	**18,865,374**	**5,190**
SD	0.144	835,195	5,762,346	5,843,195	6,061
CV	0.167	0.137	0.279	0.310	1.168
Max	1.233	7,475,402	35,060,076	31,445,142	27,289
Min	0.661	4,729,670	10,853,980	7,581,005	47
	Seepage Out (m^3/year)	**Model Pumped Water In** (m^3/year)	**Evapo-transpiration** (m^3/year)	**Gate Overflow Out** (m^3/year)	**Gate Overflow In** (m^3/year)
Average	**172,370**	**494,872**	**5,357,061**	**4,175,917**	**1,323,533**
SD	65,028	705,769	810,766	6,870,082	3,518,772
CV	0.377	1.426	0.151	1.645	2.659
Max	336,203	2,651,100	7,179,765	34,617,223	18,470,795
Min	64,583	0	4,220,988	0	0

Three of the nine modelled components in the WB had years with zero magnitudes: modelled pumped water, gate overflow out and gate overflow in. Consequently, these three components also had the largest coefficients of variation. The gate overflow components of the WB are unique in their timing and magnitudes. They occur relatively infrequently in the forty year period, but represent a substantial component of the WB when overflow arises. The modelled overflow into the Marsh from Lake Erie only occurs on 301 days (2.06 % of the total modelled days). Similarly, overflow from Big Creek over the gate into Lake Erie was modelled 678 days (4.64 % of the total modelled days). Modelled pumping has a high coefficient of variation because of the limited pumping season in a calendar year, as well as its redundancy in years with high precipitation and streamflow, reducing or eliminating the need for pumping in those years.

Seepage into Big Creek, the least significant flow in the WB, also had a large spread of data (CV = 1.168) with a minimum estimated inflow of nearly zero (just 47 m^3 in a single year). The other components of the WB had much smaller dispersions of their annual values with the coefficients of variation ranging from 0.137 to 0.377. The average water level in Big Creek Marsh is 0.862 m, which is higher than expected since the average target depth under the hemi phase of operations is 0.633 m. However, winter and early spring operations (January – April) account for the majority of the above average monthly water levels. The water levels in the Marsh during frozen conditions are not necessarily of concern to operators. Monthly and annual statistics are outlined in the appendix.

6.3.1 Marsh Storage

The following section contains a short description of the Marsh's temporal storage variation over the study period. Figure 6-6 provides the model variation of total daily storage in Big Creek Marsh over the forty year study period. There are significant fluctuations in the modelled storage with a mild seasonal trend related to the wetland's target water level. From a visual inspection of the figure, in the most recent 10 years of the modelling, both the storage peaks and the average storage level appear to be lower than their historic counterparts. These comparatively lower volumes are worth noting because the model was developed using data from this time period.

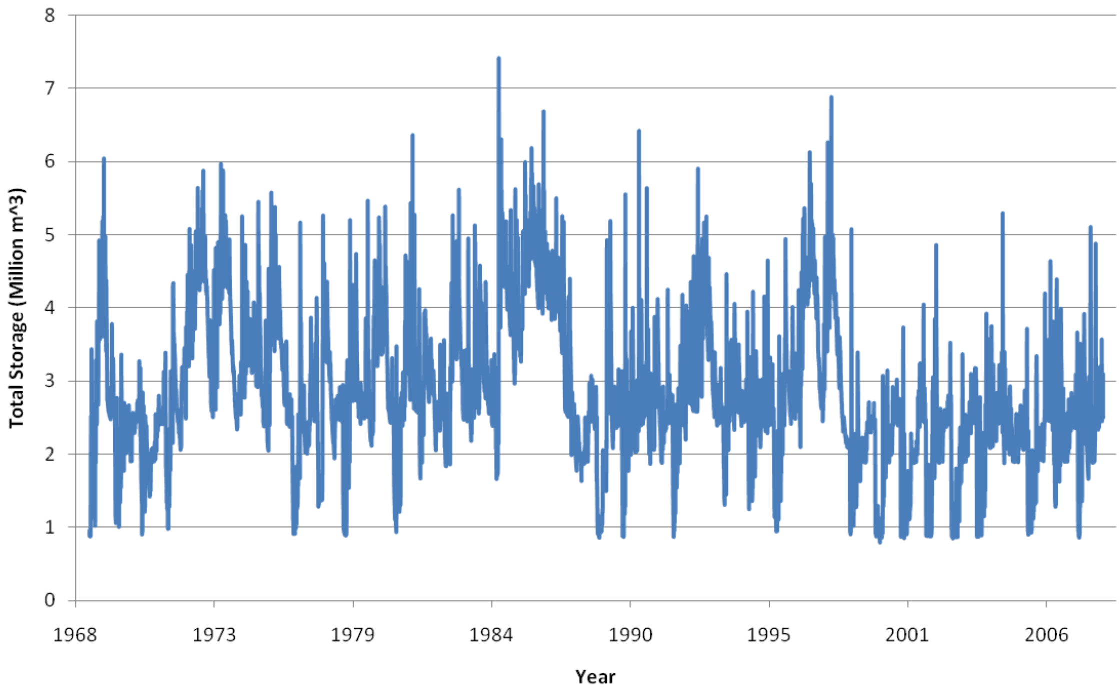

Figure 6-6: Total Storage in Big Creek

The largest modelled storage on a single day occurred on February 24, 1985 with an approximate volume of 7.4 million m^3. However, the year with the maximum average annual modelled storage was 1986 with an estimated average Marsh water level of 1.233 m and an average volume of 4.6 million m^3. The record high value in 1985 occurs because of the torrent of streamflow and precipitation the Marsh received in a single storm event. The year 1985 is in the 87.5 percentile for annual average streamflow and precipitation. A percentile is the value of a variable below which a certain percent of observations fall below, where a higher value indicates a rarer event. A unique feature contributing to this flooding year was the high water levels in Lake Erie, where the annual average Lake Erie water level was in the 95 percentile. The 1986 year had similar conditions of high inflow, high precipitation, and high water levels in Lake Erie with these parameters in the 100th, 85th, and 100th percentile, respectively. Conversely, the year 2000 had the lowest average annual storage value, falling in the 7.5th percentile for streamflow and 70th percentile for precipitation. The low total streamflow in 2000 is partly a function of the low precipitation in 1999 (falling in the 7.5th percentile).

Table 6-3 outlines monthly average storage depths in the Big Creek Marsh. The monthly average value is determined by calculating the daily average in each month for each year. Table 6-3 also provides the standard deviation, coefficient of variation, maximum and minimum values for the same data set. The target hemi phase depths are also included in Table 6-3. In the table the average water levels in the Marsh are generally higher than the target depths. From January through May, the water levels are significantly higher than the target. From June through December, the targets are still over, but the relative difference in depth is smaller.

Table 6-3: Summary Statistics of Monthly Average Storage Depths (m) during 1969-2008

Month	Average	SD	CV	Max	Min	Hemi Target
January	0.725	0.216	0.298	1.193	0.457	0.400
February	0.731	0.233	0.318	1.192	0.421	0.400
March	0.807	0.258	0.320	1.242	0.401	0.400
April	0.893	0.232	0.260	1.280	0.498	0.500
May	0.919	0.201	0.218	1.267	0.516	0.600
June	0.955	0.198	0.207	1.357	0.682	0.700
July	0.918	0.190	0.207	1.322	0.691	0.700
August	0.882	0.161	0.183	1.244	0.689	0.700
September	0.902	0.124	0.138	1.200	0.781	0.800
October	0.863	0.086	0.100	1.248	0.755	0.800
November	0.866	0.096	0.111	1.191	0.739	0.800
December	0.884	0.090	0.101	1.232	0.732	0.800

6.3.2 Sensitivity Analysis

This section contains an investigation of the WB model's sensitivity. To review the relative effects of each WB component, a series of single variable sensitivity analyses were performed on select model inputs. It should be noted that the streamflow and precipitation parameters are independent from the other variables in the WB calculations (see Equation 4.1). Seepages, gate overflow, pumping, gate outflow, ET and water storage are variables dependent on each other. The magnitude of the potential ET is an independent value from the model calculations. To calculate the model's outflow for ET, the potential rate is multiplied by the surface area of the modelled open water surface.

The variables considered for the sensitivity analyses were streamflow, potential ET (PET) (not modelled ET outflow), and hydraulic conductivity. Figures 6-7, 6-8 and 6-9 summarize the results of the sensitivity analyses. The figures outline the percent difference change in the model dependent variables to a percent difference change in a model input parameter. The average annual values are utilized in determining the percent change.

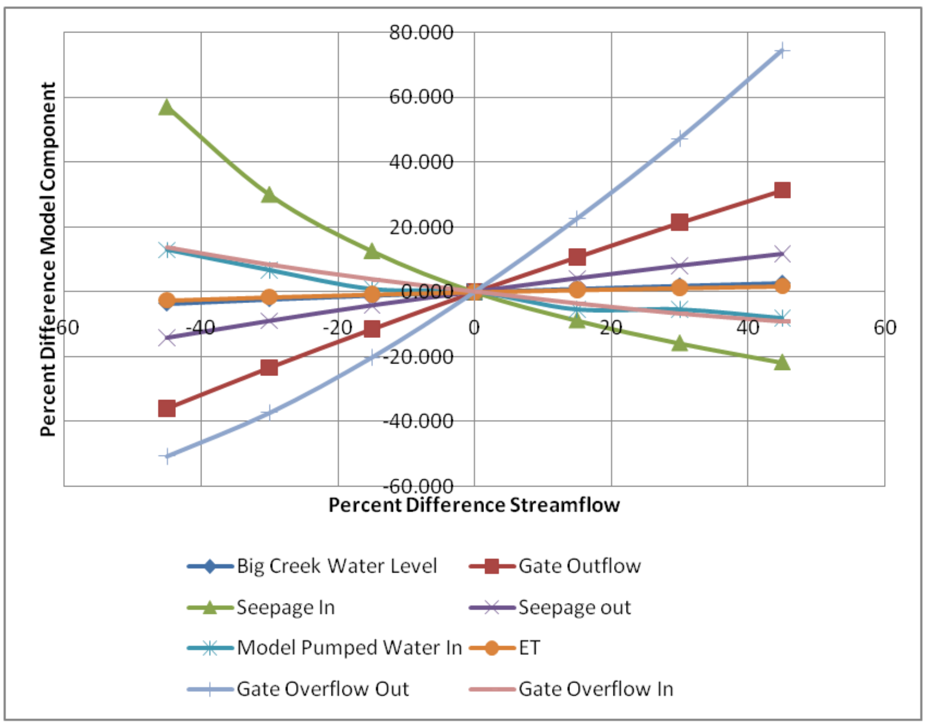

Figure 6-7: Sensitivity Analysis of Streamflow

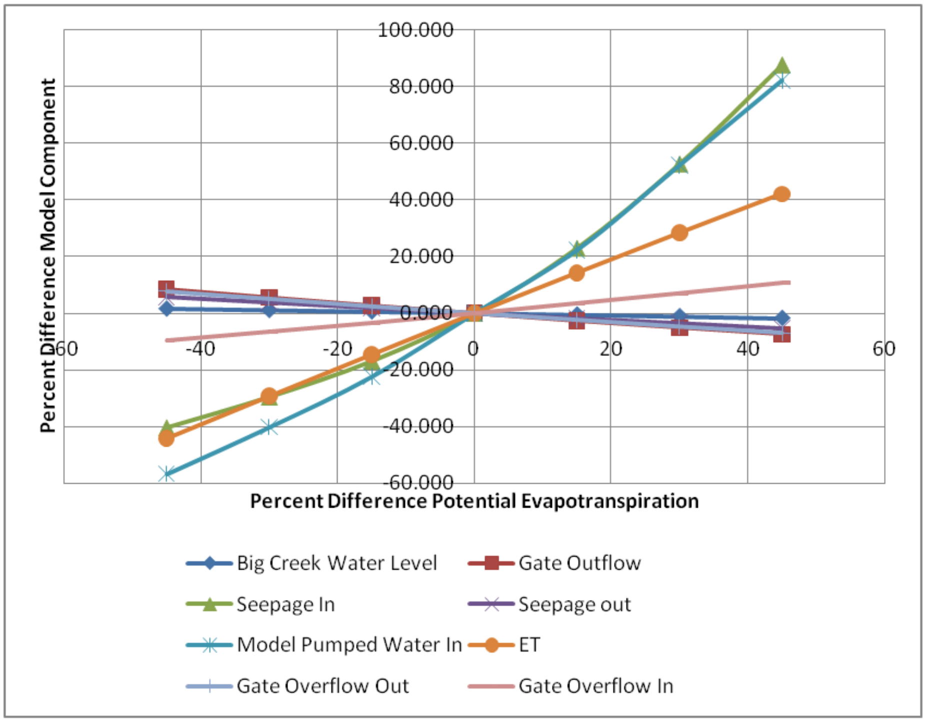

Figure 6-8: Sensitivity Analysis of Potential Evapotranspiration

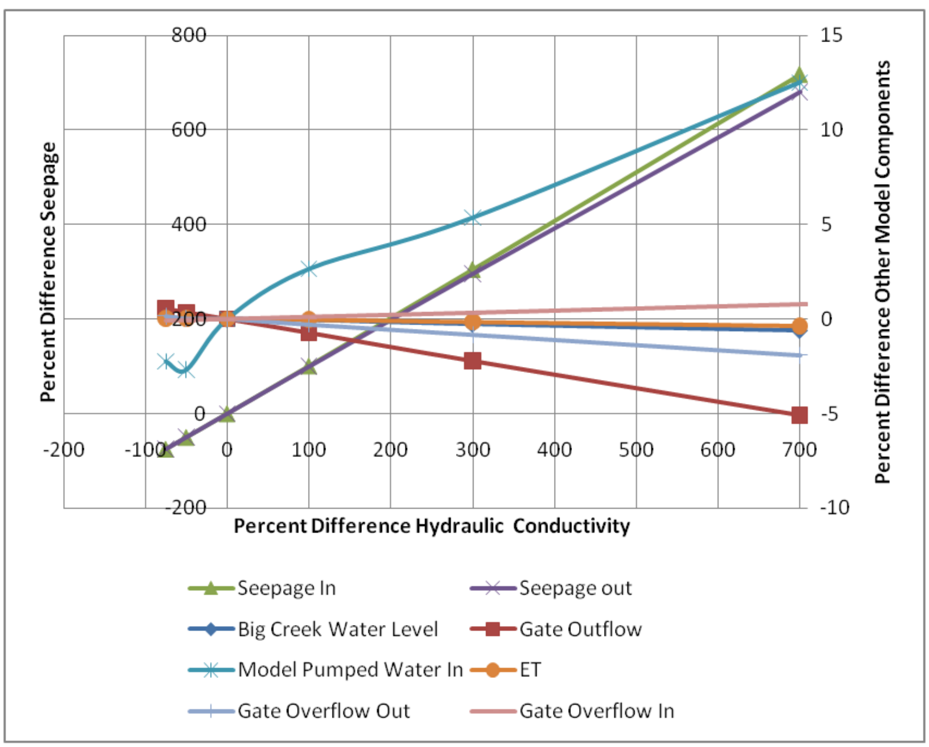

Figure 6-9: Sensitivity Analysis of Hydraulic Conductivity

For the streamflow sensitivity analysis, the default streamflow values were modified in 15% intervals from 55% of the original streamflow to 145% of the original streamflow (-45 to +45 percent difference). The effect of the modified streamflow values are highlighted in Figure 6-7. In terms of relative change in initial value, the variables most sensitive were found to be gate overflow out and seepage in. The gate overflow out is very sensitive to both positive and negative percent changes in streamflow, whereas the seepage in is more sensitive to negative percent changes. Gate outflow was moderately sensitive to both positive and negative changes in streamflow. Seepage out, gate overflow in and modelled pumped water in were mildly sensitive to changes in streamflow. ET and Marsh water level are not sensitive to alterations in streamflow.

In the PET sensitivity analysis, the default input values were inputted under the same range as the streamflow analysis (55% to 145% change). The PET is an independent model quantity that is used to calculate model ET outflow. The model ET outflow is a function of both the water in the Marsh's storage and PET. The seepage in and model pumped water were found to be the most sensitive to changes in the PET rates. The model's ET outflow component had nearly an equivalent linear slope to the input change, but in the positive percent change region ET had a minor reduction compared to the percent change. Gate overflow in was mildly sensitive to changes in the PET rate. The remaining model variables were relatively insensitive to PET changes, with the modelled Marsh water level being the least sensitive variable.

For the sensitivity analysis of the model's hydraulic conductivity, a different range was considered since this parameter's influence is smaller by orders of magnitudes. Hydraulic conductivity is proportional to the modelled seepage flow rate (see Equation 4-4). Since the inflow and outflow seepages were the least significant components of the WB (see Figures 6-4 and 6-5), a larger range of conductivities was examined. With the default hydraulic conductivity at 0.20 mm/s, the sensitivity analysis considers a percent difference range of -75% to 700% (0.05 mm/s to 1.6 mm/s).

In Figure 6-9, the primary vertical axis outlines the percent difference change in inflow and outflow seepage, whereas the secondary vertical axis represents the percent difference change in the other WB components. The seepage values were drastically affected by the change in the conductivity. The changes in both the inflow and outflow were nearly equivalent for each alteration in conductivity. The percent change in the seepage values was nearly linear. The other components of the WB were generally insensitive to changes in hydraulic conductivity, though the water pumped into the Marsh and gate overflow parameters were mildly sensitive to extreme percent changes.

6.3.3 Lake Erie's Influence on Big Creek Marsh's Water Level

The major focus of this section will be centered on Lake Erie's and Big Creek's hydrologic relationship. The modelled WB's direct exchange components between the two bodies include: gate controlled release outflow, inflow pumping, dual directional seepage, and dual directional gate overflow. The following analysis between the two bodies was conducted over the study period and is based on the WB model results only.

The average daily water level in the Marsh was 174.562 m AMSL whereas the average water level in Lake Erie was 0.205 m lower at 174.357 m AMSL. This suggests that the Marsh generally drains into the Lake, however; the opposite is possible and was simulated in the WB model. Figure 6-10 outlines the difference between the daily average water levels in the Big Creek WB to the recorded water levels at Bar Point Station. The difference between the two bodies is used to simplify the below graph, and to highlight common trends. From a visual inspection of the graph the difference between the two water levels in the most recent ten years appears to be greater than older data. A weak (R^2= 0.1049) positive linear relationship between the water level differences exists. The trend line indicates one of three things: that the average water level in the Lake is decreasing, the average water level in the Marsh is increasing or both. If the average water level difference is increasing this could be caused by an increase in water entering the Marsh.

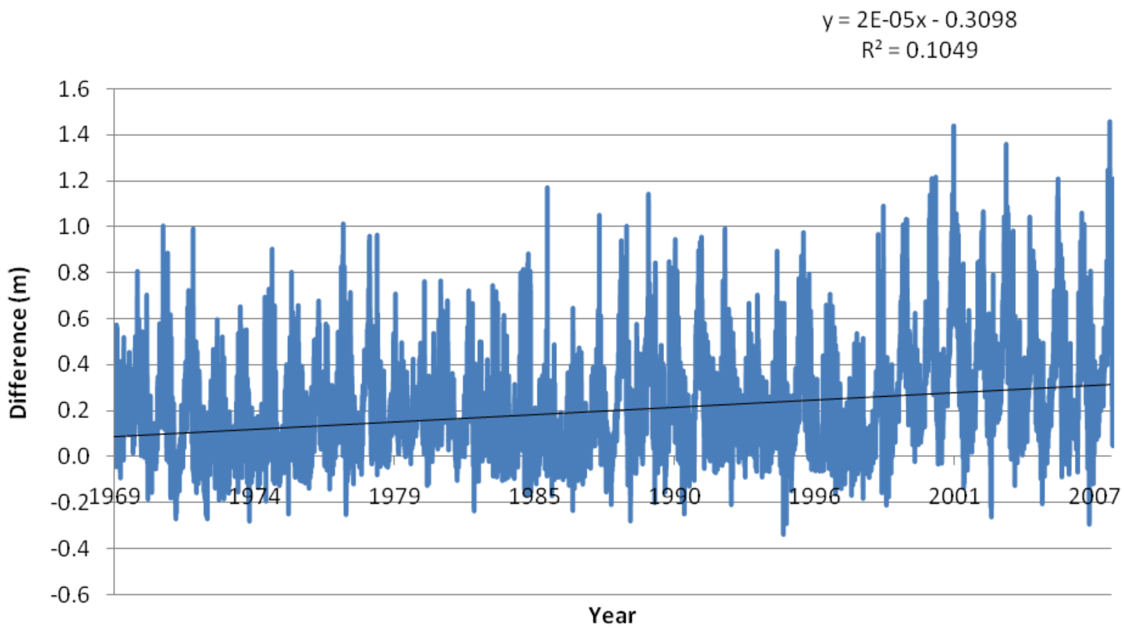

Figure 6-10: Daily Difference between Big Creek and Lake Erie Water Levels (m)

The following two tables, Table 6-4 and Table 6-5, contain additional information related to the Big Creek Marsh and Lake Erie water level relationship. In Table 6-4, the ten year average water level difference between the two bodies is outlined. The results show an increasing water level difference between the Marsh and the Lake. In the most recently analyzed decade, the average difference between the two bodies has radically increased compared to older values. From 1969 to 1978 and 1979 to 1988, the average difference is relatively stable. From 1989 to 1998, the water level difference increased slightly (approximately 27%), but the same is nearly doubled in the last decade. In Table 6-5, the annual average water levels of Big Creek and Lake Erie are summarized for each year with a corresponding percentile ranking.

Table 6-4: Ten Year Average Difference between Big Creek and Lake Erie Water Levels

Decade	Water Level Difference (m)
1969-1978	0.144
1979-1988	0.147
1989-1998	0.187
1999-2008	0.345

Table 6-5: Average Annual Water Levels in Big Creek and Lake Erie (m AMSL and Percentile)

Year	Lake Erie WL	Lake Erie Percentile	Big Creek WL	Big Creek Percentile	Year	Lake Erie WL	Lake Erie Percentile	Big Creek WL	Big Creek Percentile
1969	174.371	57.5	174.591	67.5	**1989**	174.208	27.5	174.443	25.0
1970	174.246	32.5	174.436	22.5	**1990**	174.281	40.0	174.514	45.0
1971	174.291	42.5	174.428	20.0	**1991**	174.316	45.0	174.537	50.0
1972	174.486	67.5	174.566	62.5	**1992**	174.351	47.5	174.557	60.0
1973	174.725	97.5	174.814	97.5	**1993**	174.499	75.0	174.692	82.5
1974	174.674	90.0	174.759	87.5	**1994**	174.373	62.5	174.550	55.0
1975	174.581	85.0	174.690	80.0	**1995**	174.279	37.5	174.505	40.0
1976	174.529	80.0	174.682	77.5	**1996**	174.373	60.0	174.509	42.5
1977	174.267	35.0	174.456	30.0	**1997**	174.700	92.5	174.796	95.0
1978	174.355	50.0	174.540	52.5	**1998**	174.549	82.5	174.693	85.0
1979	174.366	55.0	174.529	47.5	**1999**	174.106	12.5	174.409	12.5
1980	174.499	72.5	174.671	75.0	**2000**	173.994	7.5	174.361	2.5
1981	174.364	52.5	174.556	57.5	**2001**	173.909	2.5	174.364	5.0
1982	174.405	65.0	174.581	65.0	**2002**	174.046	10.0	174.409	15.0
1983	174.493	70.0	174.640	72.5	**2003**	173.964	5.0	174.365	7.5
1984	174.500	77.5	174.634	70.0	**2004**	174.107	15.0	174.413	17.5
1985	174.707	95.0	174.794	90.0	**2005**	174.167	25.0	174.478	37.5
1986	174.882	100.0	174.933	100.0	**2006**	174.136	20.0	174.401	10.0
1987	174.663	87.5	174.796	92.5	**2007**	174.123	17.5	174.476	35.0
1988	174.242	30.0	174.456	27.5	**2008**	174.146	22.5	174.474	32.5

Within the forty year study period, 36 of these years had the Big Creek water level percentile ranking within a 10% tolerance range of the Lake Erie percentile rank. Additionally, 27 of the years fell within a 5% tolerance range between the two bodies.

This implies that the Marsh and the Lake water levels behave similarly, which suggests that the Lake significantly and directly affects the water levels in the Marsh. While it is expected that the Lake directly affects the Marsh, the relationship between the two water bodies is also related to mutual external influencing factors (meteorological, rainfall, etc).

Streamflow and precipitation, two of the eight inflows and outflows considered in the WB model, are independent of the other six flows. All other components of the WB are indirectly or directly linked by the Lake's water level. The daily seepage flow direction and magnitude are dictated by water levels in the Lake. The daily maximum gate released flow out of the Marsh is also dictated by the Lake.

The other three flows considered in the WB (flow overtopping the dam leaving the Marsh, ET, pumped flow into the Marsh) are indirectly a function of Lake Erie's water levels. All three of these components are a function of the water volume (storage) in the Marsh. The following two figures, Figure 6-11 and Figure 6-12, compare the average annual seepage values, gate flows and pumping to the water level difference between Lake Erie and the Marsh.

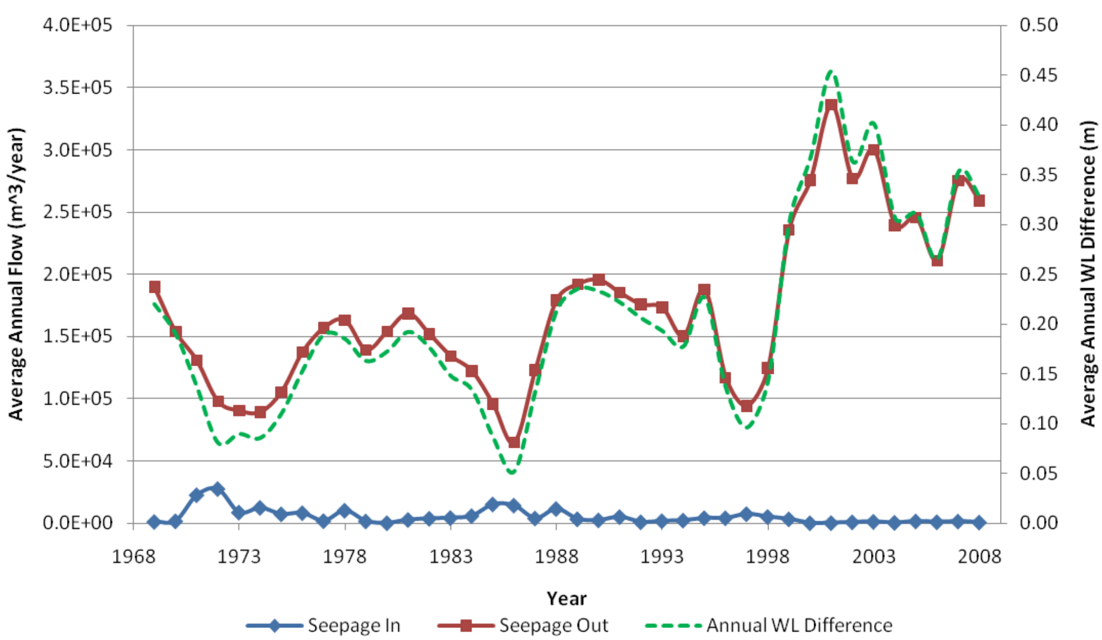

Figure 6-11: Lake Erie and Big Creek Water Level Difference's Affect on Seepage

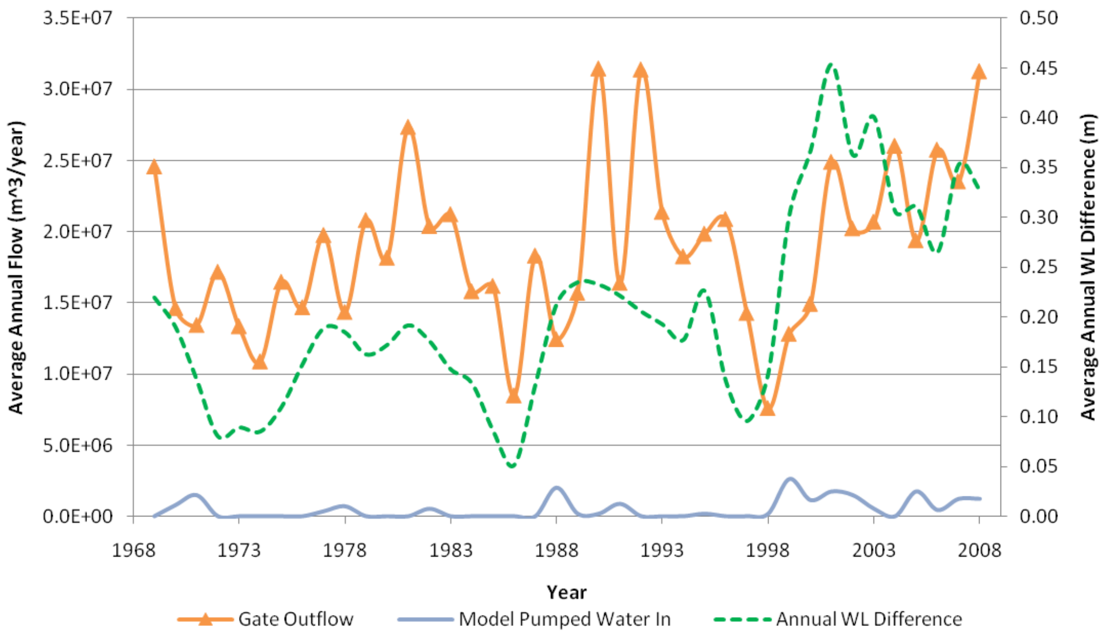

Figure 6-12: Lake Erie and Big Creek Water Level Difference's Affect on Gate Flow and Pumping

The previous two figures use the water level difference for comparison because it is a single variable that incorporates both the storage in Big Creek and the Lake's depth. From Figure 6-11, the general relationship between seepage and water level difference indicates that outflow seepage follows the water level difference trend line, and that inflow seepage increases as water level differences decreases. These trends are expected given that the equation used to model the seepage flow is dependent on the water level difference of the two bodies. Interestingly, the modelled seepage in has substantially decreased inflow to the Marsh during the most recent 10 years (1999 to 2008). Conversely, in the most recent 10 years the outflow seepage average annual values substantially increase and maintain the higher outflows. From these results, it is inferred that the wetland might experience a natural drawdown if not artificially maintained with the outlet control gate.

In Figure 6-12, the relationship between the variables is not as apparent as in Figure 6-11. From visual inspection, a correlation between the gate released outflow, inflow pumping, and water level difference can be observed. Local maximum pumping peaks generally occur around the same year that local valleys occur in the water level difference. Also, the minimum gate release flow years often occur around the same time as local valleys occur in the water level difference. An obvious relationship between the pumping and gate release flows occurs during drier years, when the gate released flow would be reduced and the pumping would be increased, as outlined in this figure.

6.4 Summary

In this chapter an investigation of the three potential operating scenarios (overgrown, hemi and open water) suggested by the Big Creek PTTW (Ducks Unlimited Canada, 2007) was undertaken. The hemi phase was found to be the most likely average operating plan of the three. This operating schedule generally fit the best with the recorded pumping data and represents the average water level goal for the wetland which would be employed the majority of the time. The open water phase has very low summer water levels, the overgrown has high summer water levels, and the hemi phase has intermediate water levels. All three operating scenarios were modelled. Both the 2006 and 2008 data fit well with the hemi scenario pumping and the overgrown phase appears to better fit the real data in 2007. The model analysis over the full forty year study period was undertaken with the assumption that the hemi phase would best represent overall historical operating practices.

Following the hemi phase operating plan, the conceptual WB for the Big Creek Marsh provides a baseline for understanding the hydrologic processes within the Marsh. The majority of individual annual WB components are well distributed including: the Big Creek water level, precipitation, streamflow in, gate released outflow, seepage out, and ET. However, because of the model's complexity, daily influxes, and seasonal trends, the same cannot be said about the distribution of the daily flows. The daily values of the

WB components are not normally or well distributed. The data's normality was assessed simply by reviewing the data set's mean, median, mode, and standard deviation relationships.

The results of the WB found the most significant inflow into the marsh to be streamflow, followed by precipitation, flow overtopping the gate entering the Marsh, pump inflow and seepage inflow. The most significant outflow was gate released outflow, followed by ET, gate overflow out and seepage outflow.

The relative timing, magnitude and importance of each WB component varied considerably. The daily magnitude of ET is small compared to other variables, but this variable is one of the most persistent. Considering the long term effect, the annual ET outflow becomes an important WB component. Gate outflow is a highly variable component, representing the majority of the modelled outflow. Pumping into the Marsh only occurs within select months; wet year pumping was not modelled. The pumping component represents a comparably small inflow, but is significant because it allows the Marsh to sustain the target water levels in drier years.

The seepage in and out of Big Creek represented the smallest annual average WB inflow and outflow components, respectively. The inflow seepage was a trivial component in each of the study years, with a maximum annual rate less than half of the minimum annual seepage out rate. The outflow seepage had a more pronounced effect in the most recent 10 years of the model; however, it still represents a minimal effect on the Marsh. The seepage outflow's effect in these years may be partially responsible for the modelled increase in pumping in the same time period.

The modelled inflow and outflow overtopping the gate occurs sporadically, but the comparatively large magnitude of these components make both of these components significant processes in the Big Creek WB. These flows are both a function of water levels in the different bodies. Only limited data existed related to the Marsh control gate (specifically the top elevation of the gate). As a result, assumptions were made to determine overtopping flow characteristics. These assumptions could represent potential error in the modelling process.

A simple review of Lake Erie's influence on Big Creek Marsh's hydrology was undertaken. A strong relationship between the two bodies' average annual water level depth ranking was outlined. This robust relationship inferred that the Lake significantly and directly affects the water levels in the Marsh, though other mutual influencing factors may also impact this relationship, especially meteorological dynamics. The difference of the average annual water levels correlate extremely well with the seepage data and moderately well with pumping and gate released outflow.

CHAPTER 7: CONCLUSIONS AND RECOMMENDATIONS

7.1 Conclusions

This study was conducted to meet two major research objectives. The first major research objective was to investigate the non-point source (NPS) loadings and hydrology in the Big Creek Watershed using the AnnAGNPS model. The second major objective was to investigate and quantify the hydrological processes within Big Creek Marsh.

7.1.1 AnnAGNPS

The Big Creek Watershed only has a limited period of continuous data which is insufficient to calibrate the model. The AnnAGNPS model was first calibrated and validated in the neighboring Canard River Watershed. The limited data from Big Creek Watershed was used to verify the AnnAGNPS model in the Big Creek Watershed. Two eight year periods, a calibration period and validation period, were utilized to assess the AnnAGNPS model's applicability in the Essex Region. The model's prediction of monthly streamflow was generally good to very good in the Canard River Watershed. The monthly Nash-Sutcliffe efficiency and coefficient of determination for the calibration period were 0.746 and 0.751, respectively. The monthly Nash-Sutcliffe efficiency and coefficient of determination for the validation period were 0.724 and 0.725, respectively. Eight months of i streamflow from a gauging station within the Big Creek Watershed had a relatively weaker match with the AnnAGNPS predictions. The predictions from the Big Creek Watershed were fair to good with a monthly Nash-Sutcliffe efficiency and coefficient of determination of 0.470 and 0.542, respectively.

The average monthly trends in the Big Creek Watershed corresponded closely with the monthly trends in the Canard River Watershed. An extensive summary of the Big Creek AnnAGNPS model outputs were outlined in Chapter 5. In general the regions producing a higher sediment and nutrient yield were located in the north-eastern and south-eastern regions of the watershed.

7.1.2 Big Creek Marsh Water Budget

An investigation of the three potential operating scenarios was reviewed. The hemi phase was found to be the most representative of an average operating plan. This operating schedule generally fits the best with the recorded pumping data and represents the average water level goal for the wetland which would be implemented the majority of the time.

Utilizing the hemi phase operations as a model guideline, a conceptual forty year water budget (WB) of the Big Creek Marsh was modelled. The most significant inflow into the marsh was found to be streamflow with precipitation as the second largest, followed by flow overtopping the gate entering the Marsh, pump inflow and seepage inflow. The most significant outflows were gate released outflow, evapotranspiration (ET), gate overflow out and seepage outflow. The seepage in and out of Big Creek represented the smallest annual average WB inflow and outflow components. The seepages represented a nearly trivial flow component of the WB.

A sensitivity analysis was conducted on the WB model components. The variables found to be the most sensitive to changes in streamflow were gate overflow out and seepage in. In the sensitivity analysis changes in the potential evapotranspiration (PET) estimates affected seepage in and model pumped water. The seepage values were drastically affected by variations in hydraulic conductivity, but other model components were not sensitive to this variable.

7.2 Recommendations

Similar to the conclusion section of this chapter, the recommendation section will be subdivided into an AnnAGNPS portion and a Big Creek Marsh WB portion.

7.2.1 AnnAGNPS

The AnnAGNPS model generally produced fair or better results with its streamflow predictions. The areas that relatively generate more sediment and nutrients were highlighted in Chapter 5. Though the exact quantity of NPS pollutant being generated cannot be verified without continuous sediment or nutrient concentration data, several conclusions can be drawn from this modelling study. Areas where remedial landuse management action would be most beneficial have been highlighted. In addition with the streamflow calibration, validation, and verification in two watersheds, it could be confirmed that AnnAGNPS can reasonably predict streamflow in the Essex County Region. However, it is suggested from this study that any modelling exercise executed with AnnAGNPS should be calibrated with streamflow data to help increase the accuracy of predictions. This is evident with the AnnAGNPS database's good efficiency match in the calibrated Canard River Watershed, but only fair match in the Big Creek Watershed.

In future research with the AnnAGNPS model, an investigation that continues testing the model's adequacy in the Essex Region could include model calibration and validation using observed continuous sediment or nutrient data. This would allow for verification of alternative model outputs which have not currently been investigated in the Ontario Region.

7.2.2 Big Creek Marsh Water Budget

The WB model provides a baseline for understanding the hydrologic processes within the Marsh. In general the WB results predict high water levels compared to the operating phase`s prescribed plan. To better predict the wetland's hydrological function, additional bathymetric data could be collected to confirm the older datasets that were used in determining the Marsh storage function. However, the predictions from the model should be considered reasonable given the general fit with the modelled and observed pumping. Further research should investigate the collection of additional wetland WB data to improve model accuracy. This could include a soil survey, observed flow observation for model calibration, or additional historic records of Marsh operations.

The results of the WB show that the seepage interaction between Lake Erie and the Marsh represents a small portion of the total WB. However, the water levels in the Lake do significantly affect the hydrological conditions of the Marsh. The rate of which outflow from the Marsh can occur is a function of the Lake`s water level. In addition flow overtopping the Marsh's control dam in both directions was determined to be a significant component of the annual WB on years when overtopping occurred.

From the limited data, observed pumping, it is suggested that the WB model reasonably estimates the flows entering and leaving Big Creek Marsh, but it is strongly suggested that additional data is collected to verify the current results. The current WB model frame work could be investigated in other wetlands where surface water flows account for the predominate hydrological processes.

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APPENDIX

Table A- 1: Sky Cover Conversion

Textual Description	Percent Cloud Cover
Clear	50
Cloudy	90
Drizzle	60
Fog	90
Freezing Rain	73
Haze	80
Mainly Clear	40
Moderate Rain	65
Mostly Cloudy	85
Rain	70
Rain Showers	78
Snow	75
Snow Showers	79
Thunderstorms	75

Table A- 2: Annual Streamflow Comparison

Year	Precipitation (mm)	Observed Streamflow (mm)	Predicted Streamflow (mm)	Percent Difference - Streamflow
1990	1237	640	597	6.6
1991	806	216	235	-9.0
1992	1177	459	505	-10.1
1993	814	316	309	2.0
1994	873	221	277	-25.6
1995	982	265	319	-20.0
1996	954	333	324	2.9
1997	952	356	329	7.7
1998	836	300	285	4.8
1999	759	158	182	-15.1
2000	1077	162	330	-103.7
2001	865	344	381	-10.8
2002	817	276	254	7.8
2003	822	227	233	-2.6
2004	998	392	326	16.8
2005	823	323	282	12.9

Table A- 3: Un-calibrated Curve Numbers

Curve Number ID:	CN "A"	CN "B"	CN "C"	CN "D"
Small Grain Straight Row Poor	65	76	84	88
Crop Residue Cover	74	83	88	90
Rangeland Poor	65	80	87	93
C1	40	67	78	86
C2	65	76	84	88
C3	69	82	85	89
C4	75	92	92	92
C5	81	93	93	93
C6	85	94	94	94
C7	89	95	95	95
Buffer	30	85	71	78
Urban Pervious	81	88	91	93
Urban Impervious	83	89	92	93
Pasture (Poor)	68	79	86	89

Table A- 4: Calibrated Curve Numbers

Curve Number ID:	**CN "A"**	**CN "B"**	**CN "C"**	**CN "D"**
Small Grain Straight Row Poor	69	78	83	87
Crop Residue Cover	71	78	86	87
Rangeland Poor	65	74	85	87
C1	30	55	70	75
C2	44	63	76	82
C3	53	68	78	81
C4	52	69	78	82
C5	58	72	80	84
C6	61	75	83	85
C7	74	83	88	90
Buffer	30	47	64	72
Urban Pervious	81	85	89	91
Urban Impervious	86	91	92	92
Pasture (Poor)	79	79	79	81

Table A- 5: Monthly Water Budget Averages

	Head (Big Creek)	**Target Head**	**Gate Overflow OUT**	**Gate Overflow IN**
month	m AMSL	m AMSL	m^3/month	m^3/month
Jan	174.425	174.1	198,322	23,308
Feb	174.430	174.1	336,748	67,503
Mar	174.507	174.1	644,704	145,848
Apr	174.593	174.2	540,768	278,803
May	174.619	174.3	373,045	251,735
Jun	174.655	174.4	596,310	194,952
Jul	174.618	174.4	441,167	137,972
Aug	174.582	174.4	166,642	51,504
Sep	174.602	174.5	306,899	6,048
Oct	174.563	174.5	223,502	56,474
Nov	174.566	174.5	114,385	35,566
Dec	174.584	174.5	233,425	73,819
Sum			4,175,917	1,323,533

Table A- 6: Flow Rate Though Control Dam

Water Level Difference Between Big Creek Marsh and Lake Erie (m)	**Time Required to Gravity Drop Water Through the Dam (days)**	**Time Required to Gravity Drop Water Through the Dam (hrs)**	**Average Q m^3/sec**
0.90 to 0.85	0.52	12.5	7.63
0.85 to 0.80	0.57	13.7	6.9
0.80 to 0.75	0.63	15.1	6.28
0.75 to 0.70	0.69	16.6	5.75
0.70 to 0.65	0.76	18.2	5.23
0.65 to 0.60	0.8	19.2	4.7
0.60 to 0.55	0.9	21.6	4.15
0.55 to 0.50	1.1	26.4	3.6
0.50 to 0.45	1.2	28.8	3.1
0.45 to 0.40	1.4	33.6	2.65
0.40 to 0.35	1.6	38.4	2.23
0.35 to 0.30	2.1	50.4	1.76
0.30 to 0.25	2.5	60	1.42
0.25 to 0.20	3	72	1.14
0.20 to 0.15	4.6	110.4	0.73
0.15 to 0.10	8	192	0.4
0.10 to 0.05	15.6	374.4	0.2
0.05 to 0.00	61.1	1,466.40	0.05

Table A- 7: Big Creek Marsh Monthly Precipitation (mm)

Month / Year	January	February	March	April	May	June	July	August	September	October	November	December
1969	85.1	8.6	55.0	108.1	128.3	114.0	232.1	87.3	45.7	54.0	89.7	63.9
1970	33.1	34.6	68.3	97.3	79.8	76.6	116.0	20.2	53.8	47.2	74.8	55.3
1971	27.5	106.0	62.3	29.0	41.0	75.8	47.9	66.2	65.4	27.1	54.3	113.4
1972	46.9	32.2	83.6	105.3	54.9	63.1	51.7	142.9	96.4	62.8	112.4	97.9
1973	41.6	37.4	133.7	39.5	91.7	129.2	100.4	33.6	71.9	53.8	87.1	90.5
1974	69.8	56.9	111.7	63.9	90.3	52.4	52.4	28.1	50.7	20.6	100.9	110.5
1975	77.5	75.5	65.0	67.9	47.4	83.5	44.4	192.4	89.1	30.5	50.4	101.8
1976	67.2	143.8	111.3	73.9	95.3	89.9	55.0	22.0	111.4	62.8	20.4	24.5
1977	17.7	49.3	106.5	129.2	52.0	92.6	77.8	109.6	153.1	48.8	71.6	87.1
1978	90.4	6.4	80.4	96.3	97.4	73.4	39.1	29.3	70.9	68.3	76.3	77.3
1979	39.2	16.1	74.3	133.1	105.0	69.2	70.0	62.0	41.2	64.2	113.7	81.3
1980	25.3	26.9	105.5	79.4	81.8	135.2	117.4	154.2	96.8	50.0	25.3	67.2
1981	12.8	76.3	28.7	114.2	70.9	125.2	157.9	65.1	212.2	99.3	34.5	72.2
1982	78.6	36.2	84.0	58.4	47.3	65.6	80.6	25.6	74.4	28.6	160.6	88.2
1983	24.4	32.3	67.4	114.5	144.5	97.5	127.7	50.3	59.3	85.1	127.2	104.8
1984	20.5	42.4	95.5	70.4	105.7	63.3	32.9	88.5	79.8	58.5	73.2	78.9
1985	51.1	120.9	124.1	41.1	88.3	55.5	76.7	125.2	62.5	98.9	176.2	42.5
1986	26.5	95.1	58.6	76.1	63.7	145.8	70.1	103.4	213.3	96.9	64.3	82.6
1987	65.0	13.3	69.2	59.3	67.5	134.8	96.4	157.0	105.2	60.5	87.7	119.5
1988	28.9	47.3	31.9	51.7	22.2	25.1	80.8	66.6	78.5	92.9	111.6	56.2
1989	33.0	18.8	44.3	75.0	144.6	121.6	64.4	62.8	69.3	64.4	64.1	34.8
1990	46.1	139.1	52.9	75.6	104.0	84.7	54.3	97.8	133.3	104.7	61.0	142.7
1991	40.5	33.7	53.0	97.5	125.4	33.1	21.1	75.3	28.5	135.8	73.3	63.6
1992	45.9	53.8	93.4	99.8	52.6	55.3	155.0	95.6	181.5	61.1	127.5	61.8
1993	94.4	40.5	81.9	77.6	46.6	109.6	54.1	41.2	115.7	56.6	43.3	23.7
1994	60.3	34.2	72.9	112.0	29.1	100.0	52.9	127.7	57.7	48.3	61.7	79.8
1995	69.7	19.4	52.7	102.0	104.0	55.4	138.3	87.6	38.8	103.4	77.1	30.4
1996	46.5	46.5	61.4	95.0	49.8	84.6	71.5	23.5	158.3	55.8	61.4	75.4
1997	67.2	104.1	89.7	40.4	118.2	89.1	44.6	84.4	85.8	50.0	35.3	66.3
1998	79.3	79.4	101.0	110.7	34.7	36.3	60.5	88.4	27.3	32.5	31.5	27.0
1999	101.9	59.0	45.3	107.1	41.2	68.1	57.9	43.2	41.3	38.4	48.7	57.3
2000	35.8	39.3	43.3	101.9	115.4	136.1	80.4	104.5	84.6	84.8	46.4	85.8
2001	23.0	70.1	33.9	71.3	77.7	44.3	35.0	40.8	118.5	163.1	85.8	54.4
2002	80.8	50.5	59.1	139.0	104.5	27.8	79.7	24.4	38.6	37.7	79.8	61.0
2003	19.4	52.5	48.7	65.2	158.3	69.2	97.6	103.4	89.9	65.6	78.3	75.9
2004	53.7	21.7	106.6	34.6	188.8	69.9	80.2	104.5	20.5	58.6	91.1	57.1
2005	104.7	77.6	32.1	92.3	32.4	21.5	91.4	52.0	65.2	10.3	90.4	88.5
2006	81.8	56.8	62.7	60.5	123.3	80.6	86.4	96.1	130.9	120.1	92.5	70.9
2007	109.4	19.2	82.0	104.5	64.5	62.4	48.4	187.4	39.4	76.9	16.8	98.6
2008	47.3	100.8	110.7	37.0	45.2	179.0	103.6	14.0	154.5	34.2	109.4	92.3

Table A- 8: Proposed Pumping Schedule during the Hemi Marsh Phase

Month	January	February	March	April	May	June	July	August	September	October	November	December
Long Term Average Lake Erie (m)	173.99	173.98	174.06	174.21	174.30	174.33	174.32	174.25	174.16	174.06	173.99	173.99
Target Marsh Water Levels (m)	174.10	174.10	174.10	174.20	174.30	174.40	174.40	174.40	174.50	174.50	174.50	174.50
Target Marsh Depths (m)	0.25	0.25	0.25	0.35	0.45	0.55	0.55	0.55	0.65	0.65	0.65	0.65
Expected Water Taking by Pumping	No	No	No	No	No	Yes	Yes	Yes	Yes	No	No	No
Direction of Pumping	-	-	-	-	-	Into Marsh	Into Marsh	Into Marsh	Into Marsh	-	-	-
Contingency Water Taking by Pumping	-	-	-	-	>174.5	>174.5	>174.5	-	-	-	-	-
Direction of Pumping	-	-	-	-	Out of Marsh	Out of Marsh	Out of Marsh	-	-	-	-	-

Source: Big Creek Pumping Plan Permit (Ducks Unlimited Canada, 2007)

Table A- 9: Proposed Pumping Schedule during the Hemi Open Water Phase

Month	January	February	March	April	May	June	July	August	September	October	November	December
Long Term Average Lake Erie (m)	173.99	173.98	174.06	174.21	174.30	174.33	174.32	174.25	174.16	174.06	173.99	173.99
Target Marsh Water Levels (m)	173.98	173.97	174.05	174.00	173.90	173.85	173.85	174.05	174.25	174.35	174.35	174.35
Target Marsh Depths (m)	0.13	0.12	0.20	0.15	0.05	0.00	0.00	0.20	0.40	0.50	0.50	0.50
Expected Water Taking by Pumping	No	No	No	Yes	Yes	Yes	No	No	Yes	Yes	No	No
Direction of Pumping	-	-	-	Out of Marsh	Out of Marsh	Out of Marsh	-	-	Into Marsh	Into Marsh	-	-
Contingency Water Taking by Pumping	-	-	-	-	-	>173.90	>173.90	>174.15	>174.35	>174.45	-	-
Direction of Pumping	-	-	-	-	-	Out of Marsh	Out of Marsh	Out of Marsh	Out of Marsh	Out of Marsh	-	-

Source: Big Creek Pumping Plan Permit (Ducks Unlimited Canada, 2007)

Table A- 10: Proposed Pumping Schedule During the Hemi Overgrown Phase

Month	January	February	March	April	May	June	July	August	September	October	November	December
Long Term Average Lake Erie (m)	173.99	173.98	174.06	174.21	174.30	174.33	174.32	174.25	174.16	174.06	173.99	173.99
Target Marsh Water Levels (m)	174.30	174.30	174.30	174.40	174.50	174.60	174.60	174.60	174.50	174.50	174.50	174.50
Target Marsh Depths (m)	0.45	0.45	0.45	0.55	0.65	0.75	0.75	0.75	0.65	0.65	0.65	0.65
Expected Water Taking by Pumping	No	No	No	No	Yes	Yes	Yes	Yes	No	No	No	No
Direction of Pumping	-	-	-	-	Into Marsh	Into Marsh	Into Marsh	Into Marsh	-	-	-	-
Contingency Water Taking by Pumping	-	-	-	>174.60	>174.60	>174.60	>174.60	>174.60	>174.60	>174.60	-	-
Direction of Pumping	-	-	-	Out of Marsh	Out of Marsh	Out of Marsh	Out of Marsh	Out of Marsh	Out of Marsh	Out of Marsh	-	-

Source: Big Creek Pumping Plan Permit (Ducks Unlimited Canada, 2007)

Table A- 11: Monthly Water Budget Averages

	Head (Big Creek)	Target Head	Precipitation	Streamflow In	Stream Flow Out	Seepage in	Seepage out	Model Pumped Water IN	Actual Pumped Water IN	Head (Big Creek)	ET
month	m AMSL	m AMSL	m^3/month	m^3/month	m^3/month	m^3/month	m^3/month	m^3/month	m^3/month	m^3/day	m^3/month
Jan	174.425	174.1	369,780	2,150,043	3,027,474	315	15,938	0	0	0.725	96,452
Feb	174.430	174.1	370,633	2,496,159	2,262,032	437	12,139	0	0	0.730	116,978
Mar	174.507	174.1	501,850	2,871,632	2,142,167	461	12,182	0	0	0.807	252,476
Apr	174.593	174.2	563,801	1,930,049	1,528,194	589	9,459	0	0	0.893	490,154
May	174.619	174.3	568,566	1,583,142	1,124,024	896	7,604	0	0	0.919	711,654
Jun	174.655	174.4	561,857	1,590,673	1,002,906	673	8,115	103,835	0	0.955	865,178
Jul	174.618	174.4	546,265	973,173	668,389	759	7,256	88,370	0	0.918	891,665
Aug	174.582	174.4	542,753	818,622	697,075	571	8,465	70,696	547,681	0.882	697,709
Sep	174.602	174.5	598,557	1,233,531	1,155,823	156	15,599	231,971	125,965	0.902	529,188
Oct	174.563	174.5	445,372	1,153,225	1,273,976	135	21,002	0	0	0.863	366,810
Nov	174.566	174.5	526,282	1,576,523	1,537,833	50	25,838	0	0	0.866	210,965
Dec	174.584	174.5	504,995	2,309,066	2,445,481	147	28,773	0	0	0.884	127,831
Sum			6,100,712	20,685,841	18,865,374	5,190	172,370	494,872	673,645		5,357,061

Table A- 12: Annual Water Budget Totals (1969-1988)

year	Big Creek Water Level m	Precipitation m3/year	Streamflow m3/year	Gate Outflow m3/year	Seepage In m3/year	Seepage out m3/year	Model Pumped Water In m3/year	ET m3/year	Gate Overflow Out m3/year	Gate Overflow In m3/year	Actual Pumped Water IN m3/year
1969	0.891	7,308,312	30,890,956	24,564,383	1,076	189,716	0	5,148,750	6,608,510	0	0
1970	0.736	5,160,353	13,136,290	14,616,336	1,568	153,677	795,330	4,462,033	0	0	0
1971	0.728	4,881,074	12,739,902	13,414,942	22,391	130,689	1,502,290	4,699,580	0	0	0
1972	0.866	6,477,295	17,730,052	17,158,457	27,289	97,868	0	5,126,060	897,162	0	0
1973	1.114	6,207,905	20,019,325	13,335,062	8,529	90,245	0	6,657,134	15,181,651	8,216,497	0
1974	1.059	5,509,878	17,958,822	10,863,088	12,273	89,050	0	6,359,884	13,626,020	7,132,011	0
1975	0.990	6,309,182	19,836,701	16,457,606	7,171	105,262	0	5,778,406	4,670,669	402,069	0
1976	0.982	5,982,163	18,951,046	14,670,672	8,055	137,395	0	6,663,704	4,233,893	529,175	0
1977	0.756	6,784,877	18,277,812	19,744,691	1,718	156,699	353,480	4,823,028	693,459	0	0
1978	0.840	5,490,100	15,202,168	14,334,817	10,015	163,161	706,960	5,106,471	1,373,232	0	0
1979	0.829	5,926,580	22,453,936	20,795,694	1,510	139,263	0	4,906,163	2,045,875	0	0
1980	0.971	6,577,890	21,300,270	18,149,614	87	153,874	0	5,653,900	4,701,096	0	0
1981	0.856	7,289,898	30,012,550	27,343,454	2,671	168,433	0	5,120,206	4,590,801	0	0
1982	0.881	5,645,937	19,617,122	20,380,663	3,588	151,781	530,220	5,301,252	0	0	0
1983	0.940	7,057,336	24,770,002	21,199,655	4,183	134,277	0	5,791,713	4,895,417	0	0
1984	0.934	5,520,108	17,487,905	15,794,468	5,739	122,267	0	5,253,943	1,704,648	246,914	0
1985	1.094	7,246,932	28,987,329	16,164,621	15,004	95,736	0	6,653,206	19,762,265	6,907,747	0
1986	1.233	7,475,402	25,419,714	8,470,942	14,114	64,583	0	7,179,765	34,617,223	18,470,795	0
1987	1.096	7,059,723	23,766,402	18,293,997	3,829	122,985	0	7,153,845	9,528,826	2,992,097	0
1988	0.756	4,729,670	10,853,980	12,430,733	11,402	179,126	2,032,510	5,710,468	0	0	0

Table A- 13: Annual Water Budget Totals (1989-2008)

year	Big Creek Water Level m	Precipitation m3/year	Streamflow m3/year	Gate Outflow m3/year	Seepage In m3/year	Seepage out m3/year	Model Pumped Water In m3/year	ET m3/year	Gate Overflow Out m3/year	Gate Overflow In m3/year	Actual Pumped Water IN m3/year
1989	0.743	5,434,858	17,670,154	15,677,343	3,203	191,921	176,740	4,509,852	2,483,485	0	0
1990	0.814	7,474,379	35,060,076	31,445,142	2,357	195,468	176,740	5,119,040	4,542,575	0	0
1991	0.837	5,323,351	14,158,038	16,386,071	4,928	185,047	883,700	5,470,973	0	0	0
1992	0.857	7,387,083	31,073,501	31,378,395	835	175,641	0	4,276,522	389,726	0	0
1993	0.992	5,354,041	24,932,490	21,373,877	1,670	173,144	0	4,944,001	6,395,328	362,309	0
1994	0.850	5,704,248	17,859,766	18,246,277	2,121	150,247	0	5,000,132	0	0	0
1995	0.805	5,992,734	18,868,453	19,838,212	4,045	187,647	176,740	5,028,491	171,788	0	0
1996	0.809	5,656,508	22,234,734	20,865,116	4,046	116,876	0	5,418,122	704,224	0	0
1997	1.096	5,966,477	19,610,347	14,262,371	7,259	93,924	0	6,287,720	10,860,314	5,657,952	0
1998	0.993	4,830,947	16,104,448	7,581,005	5,033	124,630	176,740	6,756,016	9,660,954	2,023,747	0
1999	0.709	4,836,403	11,684,292	12,799,646	3,158	235,422	2,651,100	5,353,865	468,996	0	0
2000	0.661	6,534,583	11,903,477	14,907,267	47	275,478	1,148,810	4,232,144	0	0	0
2001	0.664	5,576,373	22,147,134	24,882,683	134	336,203	1,767,400	4,452,284	0	0	0
2002	0.709	5,338,355	19,210,706	20,211,951	748	276,843	1,502,290	4,995,442	310,154	0	0
2003	0.665	6,299,634	18,126,192	20,674,866	1,157	300,052	530,220	4,220,988	0	0	0
2004	0.713	6,049,681	25,391,882	26,013,970	345	239,202	0	4,282,704	0	0	0
2005	0.778	5,170,583	18,240,864	19,377,206	1,469	245,390	1,767,400	5,458,950	717,005	0	0
2006	0.701	7,243,863	23,035,296	25,743,403	862	211,048	441,850	4,655,857	167,248	0	348,826
2007	0.776	6,202,790	21,348,208	23,514,496	1,392	275,370	1,237,180	5,243,057	0	0	658,336
2008	0.774	7,010,960	29,361,283	31,251,757	584	259,152	1,237,180	5,026,770	1,034,139	0	1,013,774

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